

## JRC TECHNICAL REPORTS

# Comparative Life-Cycle Assessment of Alternative Feedstock for Plastics Production

*Draft report for stakeholder consultation - Part 2*

*- 10 LCA case studies*

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**Deadline for consultation comments: June 30, 2020**

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JRCxxxxx

EUR xxxxx xx

Print	ISBN XXX-XX-XX-XXXXX-X	ISSN XXXX-XXXX	doi:XX.XXXX/XXXXXX
PDF	ISBN XXX-XX-XX-XXXXX-X	ISSN XXXX-XXXX	doi:XX.XXXX/XXXXXX

Ispra: European Commission, 2020

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How to cite this report: Nessi S., Sinkko T., Bulgheroni C., Garcia-Gutierrez P., Giuntoli J., Konti A., Sanye-Mengual E., Tonini D., Pant R., Marelli L., *Comparative Life Cycle Assessment (LCA) of Alternative Feedstock for Plastics Production – Part 2*, European Commission, Ispra, 2020, ISBN 978-92-79-XXXXX-X, doi:10.2760/XXXXX, JRCXXXXXX.

Printed in XXX

**Administrative Arrangement**

**JRC. 34854-2017**

**DG GROW N SI2.762599**

**"Comparative Life-Cycle Assessment of Alternative Feedstock for Plastics  
Production"**

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**Status: June 3, 2020**

**Deadline for consultation comments: June 30, 2020**

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- 1 **Acknowledgements**
- 2 Will be included in the final document
- 3 ***Authors***
- 4 List of authors

- 1 **Abstract**
- 2 Will be included in the final document

## 1 Introduction

On January 2018, the European Commission officially adopted the *European Strategy for Plastics in a Circular Economy* (COM(2018 28 final; EC, 2018a). The latter proposes a vision where innovative materials are developed and alternative feedstock sources are applied for plastics production, 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 biodegradable properties provides clear environmental benefits. Therefore, the Commission has engaged to investigate the potential environmental impacts of alternative feedstock sources for plastic production, as well as to develop Life Cycle Assessment (LCA) studies to better understand the conditions under which the use of biodegradable or compostable plastics may be beneficial.

In this framework, the Joint Research Centre (JRC) has been entrusted by DG GROW with the project “*Environmental sustainability assessment comparing through the means of lifecycle assessment the potential environmental impacts of the use of alternative feedstock (biomass, recycled plastics, CO<sub>2</sub>) for plastic articles in comparison to using current feedstock (oil and gas)*”. The main purpose of this project is twofold, i.e.:

- i) elaborating a consistent and appropriate LCA-based method to evaluate the potential environmental impacts of the use of alternative feedstock sources for plastic article production, in comparison to using current fossil-based feedstock, and
- ii) demonstrating the applicability of the developed methodological framework to a number of detailed LCA case studies for ten (10) specific plastic articles.

The project was articulated into the following main steps:

- A systematic review of selected existing studies related to LCA in the field of plastics;
- The development of a draft method applicable to comparative LCAs of plastics from alternative feedstock sources;
- The selection of relevant plastic articles to be investigated in the 10 demonstrative LCA case studies;
- Conduction of 5 screening LCA case studies to test the draft method;
- The first technical stakeholder consultation on the draft method and the results of the screening case studies (held in November-December 2018);
- Refinement of the draft method, accounting for the outcome of the consultation;
- Detailed LCA analysis of 10 selected plastic articles;
- A second stakeholder consultation, further refinement of the method and finalisation of the case studies;
- Peer review of the revised method and its subsequent finalisation.

This report specifically addresses the 10 demonstrative LCA case studies, describing both their development and the respective results. The specific plastic articles investigated in each case study are reported in Section 2.

It is noted that the case studies have the main purpose of demonstrating the applicability of the devised method in providing an as much as possible consistent and comprehensive comparison among the use of alternative feedstock sources or polymers for the production of the selected plastic articles. While large room is given to the discussion and interpretation of the results, by comparing the relative environmental performance of the investigated feedstock and material scenarios, the studies are not primarily intended to provide conclusive indications in support of specific strategic decisions or policies.

## 2 Assessed case studies and plastic articles

The following plastic articles were selected for assessment in the ten demonstrative LCA case studies developed in this project:

1. Beverage bottles
2. Food packaging film
3. Trays for food
4. Agricultural mulching film
5. Nursery pots
6. Building insulation boards
7. Automotive interior panels
8. Printer housing panels
9. Monobloc stacking chairs
10. Wipes

The selection was mainly made by applying a number of specific selection criteria to an initial, wider list of candidate plastic articles, as described more in detail in earlier project reports (Nessi et al., 2018). In addition, relevant suggestions and comments received during a first stakeholder consultation held in November 2018 were taken into account. The initial list of candidate articles was defined based on an extended review of market studies and data, research projects and studies, and a vast portion of the relevant scientific and technical literature identified through the systematic review performed as a first project step (see Section 1). Applied selection criteria included Policy priority, Market potential, Promise for deployment, Availability and quality of data needed to develop a LCA study, End of Life scenarios and related aspects, Article features (durability, type of application<sup>1</sup>), and Market coverage. A number of sub-criteria were also taken into account, which are not reported here, but can be found in Nessi et al. (2018).

In each case study, a number of reference and alternative scenarios relying on the use of different feedstock sources or polymers for the production of the relevant plastic article were investigated. These were defined based on dedicated market and literature surveys, as well as taking into account the inputs received during a Call for Specific Data and Information held in March 2019. The resulting combinations of material and feedstock sources selected for each plastic article are summarised in Table 2.1, while further details are available in the sections of this report addressing the single case studies.

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<sup>1</sup> Rigid or flexible application.



**Table 2.1.** Summary of the material and feedstock scenarios considered for each plastic article investigated in the LCA case studies described in this report.

Case study / Plastic article	Market Sector	Reference scenarios (fossil-based feedstock)		Alternative scenarios (plastic waste or other fossil-based feedstock)		Alternative scenarios (bio-based feedstock)	
		Material	Feedstock	Material	Feedstock	Material	Feedstock
1 - Beverage bottles	Packaging (rigid)	PET	Crude oil/ natural gas	R-PET	Waste PET (post-consumer)	Bio-PET	Sugarcane (BR; bio-MEG) Crude oil/nat. gas (PTA)
		HDPE		R-HDPE	Waste HDPE (post-consumer)	Bio-HDPE	Sugarcane (BR)
						PEF	Sugarcane (BR; bio-MEG) EU mix of starch crops (FDCA)
2 - Food packaging film	Packaging (flexible)	PP	Crude oil/ natural gas	CO <sub>2</sub> -PP	CO <sub>2</sub> (coal fired power plant) H <sub>2</sub> (average production mix)	PLA	Maize (US)
						PLA/PBAT blend	Maize (US; PLA) Crude oil/nat. gas (PBAT)
						TPS/PBAT blend	EU mix of starch crops (TPS) Crude oil/nat. gas (PBAT)
		LDPE		-	-	Bio-LDPE	Sugarcane (BR)
3 - Trays for food	Packaging (flexible)	PET	Crude oil/ natural gas	R-PET	Waste PET (post-consumer)	Bio-PET	Sugarcane (BR; bio-MEG) Crude oil/nat. gas (PTA)
		PP		-	-	PLA	Maize (US)
						TPS/PBAT blend	EU mix of starch crops (TPS) Crude oil/nat. gas (PBAT)
						Bio-PBS	Maize (US)

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Case study / Plastic article	Market Sector	Reference scenarios (fossil-based feedstock)		Alternative scenarios (plastic waste or other fossil-based feedstock)		Alternative scenarios (bio-based feedstock)	
		Material	Feedstock	Material	Feedstock	Material	Feedstock
4 - Agricultural mulching film	Agriculture	LDPE	Crude oil/natural gas	R-LDPE	Waste LDPE (post-consumer)	TPS/PBAT blend	EU mix of starch crops (TPS) Crude oil/nat. gas (PBAT)
						PLA/PBAT blend	Maize (US, PLA) Crude oil/nat. gas (PBAT)
5 - Nursery 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)
6 - Building insulation boards	Building & Construction	PUR	Crude oil/natural gas	CO <sub>2</sub> -PUR	CO <sub>2</sub> (coal fired power plant) Crude oil/nat. gas	Bio-PUR	Soybean (EU, PO) <sup>(1)</sup> Crude oil/natural gas (EO, MDI) <sup>(2)</sup>
		EPS		R-EPS	Waste EPS (post-consumer)	-	-
		-	-	R-PET	Waste PET (post-consumer)	-	-
7 - Automotive interior panels	Automotive	PP	Crude oil/natural gas	R-PP	Waste PP (post-consumer)	PLA	Maize (US)
		ABS		-	-		
		PBS		-	-	Bio-PBS	Maize (US)

Case study / Plastic article	Market Sector	Reference scenarios (fossil-based feedstock)		Alternative scenarios (plastic waste or other fossil-based feedstock)		Alternative scenarios (bio-based feedstock)	
		Material	Feedstock	Material	Feedstock	Material	Feedstock
8 - 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)
9 - Monobloc stacking Chairs	Other <sup>(3)</sup>	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)
10 - Wipes	Consumer goods	PP	Crude oil/ natural gas	-	-	Bio-PP	Sugarcane (BR)
		LDPE		-	-	Bio-LDPE	Sugarcane (BR)

<sup>(1)</sup> PO: Propylene Oxide.  
<sup>(3)</sup> EO: Ethylene Oxide; MDI: Methylene diphenyl diisocyanate.  
<sup>(3)</sup> Furniture & furniture equipment.

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### 3 Goal, general scope and limitations

This section discusses the goal of the case studies presented in this report, their overall scope and common aspects of this (e.g. impact categories, impact assessment methods, and data sources), as well as cross-cutting limitations. Specific aspects of the scope (e.g. functional unit, system boundary, specific assumptions and limitations) are instead separately discussed across the single case studies.

#### 3.1 Goal

In line with the overall objectives of the project and of the method developed in its context, the goal of the case studies presented in this report is to compare, in an as much as possible consistent and comprehensive manner, the potential environmental impacts of using alternative feedstock sources or polymers for the production of selected plastic articles with a defined, quantifiable function (Section 2).

Therefore, the studies evaluate the effects of small-scale feedstock or material substitution by specific actors within defined product supply chains (e.g. polymer suppliers, converters or brand owners) with the aim of providing support to micro-level decisions on feedstock/material selection by such actors. They do not address the effects of strategic decisions or policy initiatives implying large-scale changes (e.g. a complete substitution) in the type of feedstock or material used for the production of specific plastic articles or polymers in the EU (or in other geographical regions). For instance, the studies do not assess the possible effects of having all beverage bottles produced from specific bio-based polymers in Europe within 2030. This type of assessment would require additional considerations and a partially different approach.

#### 3.2 General scope

According to the goal outlined in Section 3.1, in the following case studies the assessment focuses on the environmental effects and impacts directly related to (the throughput of) activities and processes within the investigated product supply chains, assuming at the same time an (average) situation of equilibrium between demand and supply of the investigated feedstock and polymers. This is in line with the harmonised evaluation approach adopted in the Environmental Footprint context, and in the Method proposed in this project (Report I) to fulfil the purpose of comparing the environmental performance at the product level (as discussed above).

Since no relevant changes in feedstock or product demand/supply are considered (e.g. from strategic decisions), the assessment excludes possible effects from significant variations in scales of production (requiring for instance infrastructure changes). Any other indirect (market-mediated) effects due to relevant changes in the demand for a given feedstock or polymer are also excluded (such as possible changes in refinery outputs due to a reduced demand of naphtha for fossil-based polymer production). One notable exception is represented by the effects from Indirect Land Use Change (iLUC), which were separately assessed in addition to direct supply-chain-related impacts, to provide a more holistic, albeit uncertain comparison between the use of bio-based and fossil-based polymers for the selected plastic articles (see Section 3.4 for a broader discussion). This choice (which deliberately deviate from current provisions in PEF) was made considering the potential relevance of iLUC effects emerged from previous evaluations related to bio-fuels (EC, 2016). However, due to their higher uncertainty, effects and impacts from iLUC are addressed as additional environmental information.

More detail about specific processes and activities that, in this overall perspective, have been included in the system boundary of each product scenario, are separately reported for the single case studies in the dedicated sections of the report (Sections 4-13).

As for the geographical and temporal scope, in line with project objectives the evaluations presented in this report focus on the current (i.e. 2020) EU-average situation in terms of supply, consumption and End of Life (treatment or disposal) of the

investigated plastic articles. However, where relevant, extra-EU End of Life treatment or disposal were considered when sufficiently detailed and representative data were available to quantify the respective share. The evaluation of any future or prospective scenarios is beyond the scope of this project.

### 3.3 Impact categories and impact assessment methods

The potential environmental and human health impacts of the investigated plastic articles were assessed with reference to the whole set of default impact categories and related impact assessment (characterisation) models prescribed in the Product Environmental Footprint (PEF) context, and adopted consistently also in the present method. The full list of considered impact categories and impact assessment models is reported in Table 3.1.

The set of characterisation factors and the corresponding impact assessment method applied for each impact category is the one available in the version 2.0 of the EF reference package (Fazio et al., 2018a). The latter is the most recent version of the package that, at the time of developing this study, could be consistently and properly used in combination with the current pool of EF-compliant datasets in the LCA software applied as a support to the assessment (i.e. GaBi 9.2). Note, however, that a more recent version of the package and related methods is currently available (i.e. 3.0; Fazio et al., 2018b), and should be applied in any LCA study conforming to the present method. While methods conforming to the EF 3.0 reference package were indeed implemented in the software, they could not be properly used in combination with the pool of EF-compliant datasets, after importing the latter in a proper software database including the additional pools of datasets required for the assessment. This was due to partial discrepancies in the nomenclature of elementary flows between the imported datasets and the impact assessment methods already implemented in the database, which could not be directly fixed by the user. If applied as such, the methods would have led to wrong characterisation of specific substances, and hence to flawed impact assessment results.

The most relevant change between the two mentioned reference packages is related to characterisation factors applied to Human Toxicity and Ecotoxicity impact categories. In the most recent (3.0) version of the package, these were improved based on updated toxicological and physicochemical properties available from REACH (as described in Saouter et al., 2018). Therefore, the case studies presented in this report could not benefit of this improvement in the uncertainty level of the underlying impact assessment methods for the mentioned categories.

Consistently with the version of the applied impact assessment methods, also the normalisation and weighting factors applied in the case studies are those implemented in the EF 2.0 reference package. For the sake of clarity, they are also reported in Annex A of the present report. The set of weighting factors determined excluding the tree toxicity-related impact categories (see Table 3.1) was specifically applied, since these were not considered in the calculation of the total normalised and weighted impact score (as not yet benefitting from the abovementioned reliability improvement).

**Table 3.1.** Impact categories evaluated in the LCA case studies and corresponding impact assessment models (in line with Environmental Footprint requirements).

Impact Category	Unit	Impact Assessment Model
Climate Change, total <sup>(1)</sup>	kg CO <sub>2</sub> 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 <sub>h</sub>	USEtox model 1.01 (Rosenbaum et al., 2008)
Human Toxicity – non-cancer	CTU <sub>h</sub>	USEtox model 1.01 (Rosenbaum et al., 2008)
Particulate Matter	Disease incidence	PM method recommended by UNEP (UNEP, 2016)
Ionising Radiation	kBq U <sup>235</sup> eq.	Human health effect model (Dreicer et al., 1995; Frischknecht et al., 2000)
Photochemical Ozone Formation	kg NMVOC eq.	LOTOS-EUROS model (Van Zelm et al., 2008) as implemented in ReCiPe 2008
Acidification	mol H <sup>+</sup> 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 <sub>e</sub>	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 <sup>3</sup> 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)

<sup>(1)</sup> The indicator “Climate Change, total” is constituted by three sub-indicators: Climate Change, fossil; Climate Change, biogenic; Climate Change, land use and land use change.

### 3.4 Additional environmental effects and issues covered as “additional information”

The set of default impact categories specified in Section 3.3, cover a broad range of environmental issues for which an internationally agreed and sufficiently reliable impact assessment method is available, and which is considered mature enough to be recommended in the Environmental Footprint context. However, there are a number of

additional environmental issues that can be considered relevant for the investigated product supply chains, which go beyond those covered by the mentioned official impact categories. These issues include (but are not necessarily limited to):

- *potential biodiversity impacts* (relevant, for different aspects, to both fossil-based and bio-based plastic articles);
- additional (Climate Change) impacts from *indirect Land Use Change* (iLUC) possibly induced by the diversion of bio-based feedstock sources from their current use (e.g. food supply) to bio-plastics production; as well as
- *Littering and subsequent macro- and micro-plastic generation* at the End of Life and throughout the life cycle of (plastic) articles, with the resulting potential impacts on the environment and human health. Macro-plastic release at End of Life is especially relevant for specific categories of products such as single-use items for outdoor consumption (e.g. small-sized beverage bottles, take-away food packaging, etc.) which show a larger potential to be littered into the environment.

Due to the absence of an agreed, established or sufficiently reliable method for quantitative evaluation of these aspects, they were separately addressed in the study as additional environmental information, by means of the impact assessment methods or quantification approaches specified below.

As for *potential biodiversity impacts*, a quantitative estimate was conducted by means of an existing endpoint impact indicator expressing the potential loss of animal and vegetal species per year, due to a number of (midpoint) impact drivers, including Climate Change, Photochemical Ozone Formation, Terrestrial Acidification, Eutrophication (freshwater and marine), Ecotoxicity (terrestrial, freshwater and marine), Land Use and Water Use. The indicator is quantified through the *ReCiPe 2016* impact assessment method, relying on impact assessment models that, at least for some of the underlying midpoint impact categories, are not consistent with those applied in this method (Section 3.3). It has to be noted, however, that none of the currently available methods for quantitative biodiversity assessment (including the present one) were deemed sufficiently mature and robust to be recommended in the Environmental Footprint context.

*Potential impacts from iLUC* were evaluated limited to Climate Change effects, by means of (recalculated) GHG emission factors prescribed from the EU 2015/1513 Directive (EC, 2015), as detailed in Section 4.4.16.4 of the Method (Report I). The use of an alternative iLUC model (Schmidt et al., 2015) to estimate the Climate Change contribution from iLUC was also explored, as a sensitivity analysis, in some case studies (i.e. beverage bottles, food packaging film, building insulation boards and printer housing panels).

Finally, a quantitative estimate of the potential contribution of the investigated product supply chains to macro- and micro-plastics generation has been performed for comparative purposes, and expressed by inventory-level indicators quantifying either the loss of plastics from the techno-sphere (processes or consumers) or the release to the ocean or the terrestrial environment. The accounting framework described in Annex B of the Method (Report I) was applied, considering both macro-plastic generation at the End of Life stage (due to product littering or mismanagement of the resulting waste), as well as micro-plastic generation throughout the product life cycle (excluding End of Life). Micro-plastic generation from foreground processes within the product life cycle was estimated according to the method developed in the context of the Plastic Leak Project (hereafter referred to as "*PLP method*"; Peano et al., 2020), applied as such or further expanded with accounting approaches from the literature (Boucher and Friot, 2017) to address additional sources not covered in the former (i.e. road markings, marine coatings). Regarding, instead, macro-plastics generation at End of Life, two different approaches have been applied to assess variability of results: (i) the already mentioned *PLP method*, including both consumer's littering and waste mismanagement<sup>2</sup> (as well as, in the expanded version, the contribution from mismanagement of product waste

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<sup>2</sup> Such as direct discharge of waste to waterways, uncollected waste, poor management.

exported outside the EU), and (ii) an alternative bottom-up estimation procedure based on beach litter observations at the EU level for the relevant article (Addamo et al., 2018; Hanke et al., 2019) and apparent consumption data in the same region. As a sensitivity analysis, aggregated estimates of both micro-plastics and macro-plastics generation calculated through both versions of the *PLP method* were finally compared with estimates of the total supply-chain loss of unspecified plastic particles from foreground processes within the product life cycle, based on loss rates estimated by Ciroth and Kouame (2019) for background processes of the *ecoinvent 3.5* life cycle inventory database. The possible micro-plastics generation from plastic articles (or fragments) released at End of Life was not quantified, due to lack of complete understanding of degradation and disintegration pathways of plastics once released into the environment.

It should be noted that all the mentioned littering estimates were performed at the inventory level, in a comparative perspective. Therefore, they were not intended to provide accurate and conclusive quantifications of the actual macro- and micro-plastics generation from the investigated product supply chains.

At this stage, no impact assessment models are available nor could be developed in this project to evaluate the potential (physical or toxicological) impacts due to plastic emissions into the environment (including both macro- and micro-plastics). Therefore, these impacts could not be addressed in the following cases studies. This is mainly due to incomplete understanding of the mechanisms governing the fate, exposure, and subsequent (physical and toxicological) effects of on ecosystems and humans of plastic products released into the environment, which makes the development of a suitable impact assessment method challenging.

### 3.5 Data sources

Life cycle inventory data and datasets for foreground processes in the investigated supply chains were sourced in general agreement with the “hierarchy” prescribed in this method (and in the EF context) for the selection of secondary data(sets). No company-specific data from direct measurements at specific facilities were indeed applied in the case studies, since these were not developed by (or on behalf of) specific companies or organisations having a direct or indirect control of any production facility in the investigated supply chains.

However, a partial deviation was performed compared to the abovementioned hierarchy, in order to:

- i) ensure the use of representative and appropriate data (in terms of underlying technology, process or product) when no EF-compliant datasets are available (e.g. for the recycling process of some polymers), rather than applying EF-compliant or ILCD entry-level (EL) compliant proxies (as prescribed by the original hierarchy), as well as
- ii) to avoid excluding from the model none of the foreground processes when no EF-compliant or ILCD (EL) compliant proxy is available (as prescribed in the original hierarchy).

The use of representative datasets (e.g. for polymer recycling) was also explicitly recommended by stakeholders during the consultation process held during the development of this project. Therefore, the following amended selection criteria were applied, in hierarchical order, to source inventory data:

- Use of an EF-compliant dataset;
- Use of a representative and appropriate ILCD-EL compliant dataset from the GaBi database;
- Use of a representative and appropriate dataset from other life cycle inventory databases (e.g. *ecoinvent*);



- Use of representative and appropriate data from relevant 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 literature activity data were used, for modelling purposes these were combined with background inventory datasets selected according to the same criteria specified above for foreground datasets.

Note, however, that companies, organisations or any other supply chain actor applying the method described in this project shall apply the original data(set) selection hierarchy reported in Section 4.6 of the Method (Report I).

### 3.6 Common limitations

As in any life cycle assessment study, a number of common limitations affect the case studies presented in this report. These are acknowledged and discussed in this section. Some limitations are related to the scope of the LCA method in itself (which exclude specific aspects), while other relate to the absence of suitable methods for quantification in a LCA context. Finally, another source of limitation is due to data availability.

#### *1. Limitations associated to the scope of the LCA method*

There are a number of (environmental) aspects and impacts (some of which relevant to the investigated supply chains) which are not assessed in LCA, simply because they are beyond the nature/scope of the method.

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 or disasters (e.g. oil spills, oil fires, natural disasters etc.) are typically excluded, similarly to the burdens of improper production conditions or cultivation practices (e.g. misuse of pesticides and fertilisers in agriculture). Nonetheless, certain accidental emissions/spills that can be considered to be a structural property of the supply chain are usually accounted for in LCA studies and/or life cycle inventory datasets. As an example, in anaerobic digestion plants a certain amount of methane is leaked from flanges and through permeable membranes, and this leakage is frequently considered in LCA studies including this type of facilities. Similarly, emissions from venting of certain gases may be included in datasets for oil production, as generally are emissions from “structural” leakage during oil transport and from additional oil production to compensate for losses. Details about the inclusion or exclusion of these “structural” accidental emissions in life cycle inventories for activities relevant to the investigated supply chains are provided in the single case studies or in the documentation of the applied datasets.

In a similar perspective, LCA normally does not account for (emissions from) waste mismanagement due to incorrect citizen’s behaviour, such as littering, whose prediction is also more or less largely uncertain. However, due to the current relevance of this issue for plastic products, littering generation was addressed in the following case studies, albeit applying a preliminary accounting framework for quantification (as better discussed in Section 3.4).

#### *2. Limitations associated to the absence of a (suitable) quantification method*

Certain impact categories or environmental issues are not covered in LCA, because of the absence of a (suitable) quantification method. For instance, impacts on landscape (e.g. from oil sand extraction or land conversion to agricultural monoculture on wide areas) are not included, due to difficulties in quantifying landscape impacts and to relate them to specific process throughput. Similarly, no structured impact assessment methods are currently available to quantitatively assess the potential environmental and human-

health impacts from littering and (subsequent) macro- and micro-plastics generation (as discussed in Section 3.4).

### *3. Limitations due to data availability*

Scarcity of data, or lack of sufficiently robust, representative or consistent data represent another source of limitations for the present case studies.

While most data-related limitations are case-study specific, and are hence highlighted and reported for the relevant study in the corresponding section of this report, others are in common to all or part of the studies. One of the most relevant common data limitation relates to additive use for polymer production and during subsequent conversion into specific plastic articles. Currently available data and information is not sufficiently detailed and representative to allow a consistent and robust comparison among the use of the different investigated polymers for the selected plastics articles, and especially between conventional (fossil-based) polymers and non-drop-in bio-based polymers. Indeed, while an important effort was made by ECHA to develop a comprehensive list of polymer additives and respective average concentrations (or concentration ranges), the data only cover conventional polymers and do not differentiate between the type of application or sector (nor between food contact and non-food contact materials). Therefore, these data are not suitable, alone, to identify relevant substances used for specific polymer applications, and to perform a proper comparison at the product level. In light of this, the use of additives and the related potential impacts were only explored in one case study, namely printer housing panels (Section 11), which was considered relevant due to the possible use of flame retardants in electronics applications. For modelling purposes, available data with information from other (more outdated) sources, and by introducing additional assumptions for the selection of representative substances to be considered in the modelling itself. Due to the uncertainty associated with this assessment, it was conducted as part of the sensitivity analysis, and the respective results are presented and discussed separately.

At a more specific level, life cycle inventory data for some major foreground processes are also lacking, or are only partially available, so that suitable proxies or partial inventories had to be applied. This is, for instance, the case of production processes of some emerging intermediates, such as FDCA (Furandicarboxylic Acid) and soy-based polyols, as well as of mechanical recycling of specific polymers or products such as LDPE films, EPS boards, and post-consumer electronics plastics (including ABS and PC/ABS). In other cases, data from process simulation and process modelling had to be applied, which are usually not representative of large-scale production conditions (and do not account for possible efficiency gains due to upscaling and process improvement), despite process optimisation strategies are normally taken into account (e.g. in terms of energy integration). This is mostly the case of processes for the production of relevant intermediates to emerging or maturing polymers, such as Hydroxymethylfurfural (an intermediate to PEF production) and CO<sub>2</sub>-based methanol and olefins (precursors to CO<sub>2</sub>-based PP). These limitations are clearly acknowledged throughout the case studies, and properly taken into account in the interpretation of the respective results.

## **3.7 Value choices, normative decisions and transparency**

As many other analytical tools, LCA involves the creation of a conceptual model of reality. This artificial reduction of complex systems to a simplified model leads to inevitable value choices and assumptions. In the case of a life cycle model, these mainly relate to some methodological choices (e.g. handling of multi-functionality situations, time horizons, etc.), and to specific supply-chain configurations (e.g. types of feedstock, End of Life scenarios, etc.) when no representative data or information is available. Value choices cannot be deemed 'right' or 'wrong', and additional research and resources would not bring additional clarity on which is the best choice to make. Rather, they can be considered as a crucial part of the evidence production process.

Building largely upon existing normative provisions given in the Product Environmental Footprint context, the case studies implement several methodological choices that have been defined ex-ante to support more consistent, robust and reproducible LCA studies, based on a participatory process involving multiple and diverse stakeholders. They also account for additional choices specifically defined in the method developed for the purpose of this project, which were transparently described in the methodological guide, and discussed as well with a broad range of stakeholders throughout the project (in two open consultation processes at different project stages).

Specific assumptions, mainly linked to supply chain configurations, are transparently described within each case study and, wherever possible, assessed through sensitivity analysis to evaluate the relevance of their effects on LCA results. Examples are the choice of the feedstock (e.g. the type of crop) for polymer production, the recycled content in specific plastic articles, as well as the technical performance prioritised in the functional unit.

### **3.8 Handling of polymers relying on emerging technologies**

By its inherent nature, this project involves the assessment of products and polymers relying on production technologies that nowadays do not necessarily present the same level of development and maturity. Polymers totally or partially derived from alternative feedstock sources frequently rely on processes running at smaller scales (in a few cases not yet applied for commercial production), or that are only at an early or medium stage of development, compared to more established conventional (fossil-based) polymers. The latter have generally benefitted of many years of process optimisation (e.g. energy or process integration) and upscaling, leading to improved energy efficiency and yields, which are normally reflected by an improved environmental performance. Conversely, production processes of polymers based on alternative feedstock sources may undergo rather important optimisation and scale effects in the future, potentially leading to an improved environmental profile (in contrast to conventional polymers, which show more limited options for improvement).

This possible difference in the level of maturity and/or production scale is clearly acknowledged, where relevant, across the different case studies, and properly taken into account in the presentation and interpretation of the results. For instance, results related to articles based on polymers relying on emerging technologies (and possibly modelled through data derived from process simulation rather than from real production facilities) were separately presented from those related to polymers relying on (more) established technologies. This is the case, for instance, of Polyethylene Furanoate (PEF) beverage bottles. It is noted, however, that frequently simulation data already account for some process optimisation strategies (e.g. energy integration), making them potentially more representative of large-scale production conditions than real data from current, smaller-scale facilities.

Considering the temporal scope of this project (which focuses on present conditions, i.e. 2020) and current data (and method) availability, no prospective scenarios accounting for estimated improvements from possible process optimisation and upscaling were considered in this project. Indeed, while average ranges of so-called “learning rates” are available 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 Section 4.4.5 of the Method – Report I). This makes it often difficult to predict future learning rates (Daugaard et al., 2015), or extrapolate existing rates to newer products or industries.

Applying any learning rate from the literature (or any estimated “scaling factor”<sup>3</sup>) would thus be arbitrary, and would introduce additional uncertainty in the results of any future LCA scenario. Moreover, it would not be possible to (consistently) apply such rates or

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<sup>3</sup> Based, for instance, on historical improvements in similar but more established (conventional) production processes.

factors across the investigated polymers, because production inventories of many polymers are only available in an aggregated form. Therefore it would not be possible to apply learning rates or scaling factors to the relevant process inputs (e.g. energy) or parameters (e.g. yield) potentially affected by the expected improvement (provided that these are known for available rates and factors).

In light of this, the approach adopted in this project was to address the issue within the interpretation phase of the affected case studies, presenting a gap analysis of the impact reduction needed for scenarios relying on emerging technologies to outperform (fossil-based) scenarios considered as a reference. This will not prevent interested readers to apply learning rates to the results found in this project.

### **3.9 Opportunities for further research/improvement**

This section is pending and will be drafted only at the time of finalising the overall study and report. Additional input, data, and information may indeed be gathered during the next consultation and refinement/finalisation steps, which may help overcome existing limitations.

### **3.10 Note on the structure of the report**

The following sections of this report address the different LCA case studies on the ten selected plastic articles, describing the assessed scenarios, the main modelling details and the respective results. The content of each section was defined in the attempt to make the description of the single case studies as far as possible self-standing, so that the reader can find all the relevant information related to the development of the case study(ies) of interest just by focusing on the respective section(s). In this perspective, references across the report were minimised and some repetitions (which are acknowledged by the authors) were introduced.

## 4 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). Bottles with a size of 0.5 litres were specifically considered, due to their higher potential of leakage into the environment compared to bottles having larger sizes. These are normally used for indoor consumption, where proper waste collection is more likely to take place (at least at the European level). It is important to notice that the assessment focuses exclusively on the bottles life cycle, to evaluate the effects of feedstock or material substitution in polymer or bottles manufacturing. 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 section.

### 4.1 Assessed scenarios

A number of scenarios were analysed to explore the potential impacts associated with the use of alternative feedstock sources or materials for the manufacturing of plastic beverage bottles (Table 4.1). Polymers traditionally used for bottle production are PET and HDPE from (virgin) fossil-based feedstock sources, which were considered as reference materials for the comparison (Scenarios 1 and 2). The use of recycled PET and HDPE from separately collected, post-consumer plastic waste as a feedstock was then explored (Scenarios 3 and 4), considering estimates of the current average recycled content in bottles at the EU level, i.e. 24% for PET bottles and 16% for HDPE bottles <sup>4,5</sup>. However, since it is technically feasible to incorporate higher shares of recycled polymer in bottle production (virtually up to 100% for PET bottles and non-food grade HDPE bottles), the use of a 100% recycled content was also explored as a sensitivity analysis for both materials (Section 4.7.7.1). This allows to assess the effects of a complete substitution of the virgin material by its recycled counterpart, and to quantify the maximum potential environmental benefits or drawbacks expected from this replacement.

A partially bio-based, drop-in alternative to fossil-based PET is also 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) was explored in Scenario 5, considering Brazilian sugarcane as a feedstock for bio-based MEG. This choice is in line with the origin of bioethanol currently used by the largest bio-MEG supplier worldwide, which is located in India and relies on bioethanol sourced in Brazil. In contrast to PET, it is possible to produce fully bio-based HDPE by totally replacing fossil-based Ethylene with its bio-based counterpart. This alternative was explored in Scenario 6, considering again Brazilian sugarcane-based bioethanol as a raw material for bio-Ethylene production (being sugarcane the most widely used feedstock at present for Bio-HDPE). However, the use of

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<sup>4</sup> These values represent 2020 estimates of the penetration rate of recycled material in PET and HDPE packaging applications (typically dominated by bottles). They were calculated assuming a linear variation between the penetration rates estimated for 2014 and 2025 in the study carried out by Deloitte (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 the target, according to the requirement from the revised Directive (EU 2018/852).

<sup>5</sup> An estimate of the average recycled content in PET bottles in Europe is also reported (for 2017) by EPBP (2018), i.e. 11% (based on the ICIS and Petcore Europe Survey on the European PET Recycling Industry in 2017). However, the value estimated based on the procedure described in footnote 1 was applied in this study, to keep consistency with the procedure followed for HDPE bottles (for which no official estimates of the current average recycled content are available).

alternative types of feedstock sourced in Europe was also explored, for both Bio-PET and Bio-HDPE bottles, as a sensitivity analysis (Section 4.7.7.2).

Finally, an emerging, fully bio-based alternative to fossil-based PET was explored in Scenario 7, where Polyethylene Furanoate (PEF) is used as bottle material. This polymer, which is not yet available at the commercial scale, is seen as a promising bio-based replacement for fossil-based PET (and potentially of other packaging materials like glass), while providing better barrier and mechanical properties than PET (e.g. barrier to oxygen and carbon dioxide). 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 derived from Brazilian sugarcane-based Ethanol (consistently with the feedstock considered for partially bio-based PET), while FDCA was derived from starch-based glucose from the EU-average mix of starch crops (one of the main pilot plants for FDCA production is located in Europe). The mix includes maize (47% on starch basis), wheat (40%) and potatoes (13%) (Starch Europe, 2019). Due to the relative early stage of development of the production process of the main FDCA precursor (i.e. Hydroxymethylfurfural – HMF), only data from process simulations were available to develop a life cycle inventory of this activity. While possible process optimisation strategies are taken into account in the generation of such data (e.g. energy integration), they are not representative of large-scale industrial production (in contrast to data available for fossil-based PET considered as a reference) and do not account for possible (efficiency) improvement due to upscaling and further process optimisation. Moreover, for further downstream conversion and polymerisation steps, data for similar processes were applied as an approximation. Impact assessment results related to PEF bottles thus need to be interpreted with caution, and were not directly compared with those of the reference materials relying on technologies at higher levels of maturity. Rather, any gaps to be filled to achieve a comparable or improved environmental performance were assessed (according to the discussion reported in the Method (Report I - Section 4.4.5)).

It is noted here that another fully bio-based polymer, i.e. Polylactic Acid (PLA), could also be used for bottle production, as demonstrated by different examples that can be found on the market. Other than being of bio-based origin, PLA is also biodegradable under controlled aerobic conditions, such as in composting facilities (and, to a much lower extent, also under anaerobic conditions). However, the use of this material was not considered in this assessment, since according to a relevant stakeholder PLA is unsuitable for (0.5 litre) beverage bottles, and it can interfere with PET recycling if sorting facilities are outdated<sup>6</sup>.

Regarding End of Life, all currently viable treatment and disposal options for beverage bottles made of each specific material were considered, including mechanical recycling, incineration and landfilling. For bottles relying on emerging polymers (i.e. PEF), potentially viable End of Life options as of today were considered, taking into account relevant material properties (e.g. biodegradability), and the options currently applied to bottles made of the material they intend to replace (i.e. PET). As a base case, the impacts of each beverage bottles scenario were assessed with reference to an EU-average End of Life scenario including all the options currently applied at the EU level to bottles made of the specific material (or those that would be potentially applied to them in the case of emerging materials). For all beverage bottles scenarios, these include all the viable options reported above, which were combined as described in Section 4.4.5.1 on End of Life modelling. In addition, scenario impacts were calculated by individually considering the applications of each currently viable (or potentially viable) End of Life option. Note that, despite bio-based PET and HDPE (partially) incorporate bio-based material, they are not biodegradable, as the final polymers have the same characteristics

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<sup>6</sup> "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."

as their fossil-based counterpart ("drop-in" solutions). Therefore, biological treatment options such as composting and anaerobic digestion are not viable for these materials and were not considered in this study.

**Table 4.1.** LCA scenarios assessed for the beverage bottles case study.

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

<sup>(1)</sup> The impacts of scenarios were individually assessed for each listed End of Life option, as well as for a combination of such options reflecting as far as possible the current (real or potential) average situation at the EU level.

<sup>(2)</sup> MEG: Mono Ethylene Glycol.

<sup>(3)</sup> PTA: Purified Terephthalic Acid.

<sup>(4)</sup> FDCA: Furan Dicarboxylic Acid.

<sup>(5)</sup> The mix includes Maize (47%), Wheat (40%), Potatoes (13%), in terms of starch product equivalents (Starch Europe, 2019).

## 4.2 Functional Unit and reference flow

The main function of the studied article (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 comparable shelf life of the product" (Table 4.2).

1 **Table 4.2.** Definition of the functional unit for beverage bottles LCA scenarios.

Aspect	Description
“What” (function(s) or service(s) 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(s) or service(s))	1000 litres of beverage
“How well” (expected level of quality)	Without breaking during transport, and ensuring a comparable product shelf life
“How long” (duration/lifetime of the function or service)	One time
“Where” (location/geography of the service)	In the EU

2 The reference flow of each scenario (i.e. the amount of polymer required to fulfil the  
3 functional unit), was calculated based on estimated average masses of bottles made of  
4 the relevant material, when used for the delivery of juices and non-carbonated soft  
5 drinks (Table 4.3).

6 Regardless of the type of feedstock used for polymer production, the average mass of  
7 PET and HDPE bottles was estimated based on linear regression on measured mass  
8 values reported by Markwardt et al. (2017) for bottles with a size ranging from 330 ml to  
9 1000 ml<sup>7</sup>. The resulting average mass of 0.5 litre PET bottles was equal to 24.8 g, while  
10 for HDPE bottles an average mass of 27.1 g was estimated. It is reasonable to consider  
11 that these estimates inherently take into account the different barrier and mechanical  
12 properties of the two materials, although such properties are not explicitly considered in  
13 the calculation of the bottle mass. Indeed, real packaging masses are normally a result of  
14 a more or less long (empirical) optimisation process to meet different design constraints  
15 and functional requirements with a given material and its underlying properties.

16 For PEF bottles, no measured mass values were available, due to the current absence of  
17 commercial applications of this material. An average bottle mass was thus estimated  
18 based on potential material savings compared to the reference material that PEF is  
19 intended to replace, i.e. PET. According to stakeholders’ suggestions, the improved  
20 barrier and mechanical properties of PEF allows for an overall 20% reduction in material  
21 usage with respect to PET, corresponding to a final bottle mass of 19.8 g.

22 **Table 4.3.** Calculation of the reference flow for beverage bottles LCA scenarios.

Polymer	Bottle mass (g)	Reference flow (kg/FU)
PET (all types of feedstock)	24.8	49.6
HDPE (all types of feedstock)	27.1	54.2
PEF	19.8 <sup>(1)</sup>	39.7

(<sup>1</sup>) Based on an estimated 20% potential material saving compared to PET,  
due to improved mechanical and barrier properties.

<sup>7</sup> The following size-mass data pairs were considered for PET bottles (limiting to those used in the sectors “juice and natural soft drinks” and “grab & go”): 330 ml-21.30 g (average of 2 measured values for the “grab & go” sector); 1000 ml-35.19 g (average of 4 measured values for the “juice and natural soft drinks” sector). For HDPE bottles, the following data pairs were considered: 350 ml-22.51 g (average of 1 measured value for the “grab & go” sector; 380 ml-22.07 g (single value for the “grab & go” sector); 1000 ml-44.73 g (single value for the “juice and natural soft drinks” sector).



### 4.3 System boundary

In all scenarios, the system boundary was set in order to cover the most relevant stages and processes of the full product life cycle (cradle-to-grave perspective), as described below and depicted in Figures 4.1 to 4.7:

- Feedstock Supply<sup>8</sup> – covering extraction, transport and possible refining of crude oil and natural gas (fossil-based polymers), collection, transport and sorting of plastic waste (recycled polymers), crop cultivation (bio-based polymers), as well as transport of these feedstock sources to downstream conversion processes (e.g. naphtha cracking, polymer recycling, sugarcane fermentation, wet milling of starch crops);
- Polymer Production<sup>9</sup> – covering all the activities associated with the conversion of feedstock materials into the relevant monomer(s) and final polymer, including any transport among these activities and final transport of polymer granulate to downstream manufacturing processes;
- Article Production<sup>10</sup> – including bottle manufacturing (directly at filling plants)<sup>11</sup> through stretch-blow moulding of injection-moulded preforms (PET and PEF bottles), or extrusion-blow moulding of polymer granulates (HDPE bottles);
- Distribution – including transport of bottles from the manufacturing and filling site to retailers and from these to final consumers<sup>12</sup>;
- End of Life – covering collection, transport, recycling, incineration, or disposal of bottles after use, including any avoided processes from virgin material or energy substitution.

A different nomenclature was applied for some life cycle stages compared to the default nomenclature specified in the Method (and in the PEF framework) to make it more relevant for the investigated supply chains and the project scope. Moreover, the default “Raw Material Acquisition and Pre-processing” stage was further split into two separate sub-stages (i.e. Feedstock Supply and Polymer Production), to allow disaggregating the impacts of feedstock supply from downstream conversion processes, and hence to better appreciate any differences among the use of different feedstock sources (in line with the project goal).

Filling of bottles (including all operations at filling plants beyond bottle production) and the Use Stage were excluded from the assessment. This is because this 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 different bottle materials investigated in this study does not affect the burdens of filling activities and from activities related to the Use Stage of beverages delivered by means of bottles (e.g. any chilling activity or the use of drinking cups)<sup>13</sup>. Therefore, these activities can be excluded from a comparative assessment, as being identical for all the examined scenarios.

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<sup>8</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

<sup>9</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

<sup>10</sup> Corresponding to the default stage of “Manufacturing” specified in the Method and in the PEF framework.

<sup>11</sup> This is normally the case of PET bottles used in large filling plants, where the conversion of polymer resin into preforms and 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 bottles (although these are normally manufactured in external facilities) and PEF ones.

<sup>12</sup> While filled bottles are transported in reality, transport of the respective 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).

<sup>13</sup> An exception may be represented by CO<sub>2</sub> 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

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). These components were excluded as well from the assessment, considering its 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 are used, regardless of the material or feedstock applied for bottle manufacturing, so that the comparison among alternative scenarios is 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 exist among compared alternatives, which are not taken into account). Therefore, the Use Stage, filling activities (and the life cycle of additional packaging components) shall always be taken in full consideration for possible inclusion in a comparative LCA study on beverage bottles, paying particular attention to any differences among the compared alternatives. Moreover, such stages shall always be included in LCA studies of specific packaged beverages (e.g. bottled water), in line with system boundary requirements in the Method and in the PEF framework.

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 bottles, of the examined polymers and of plastics in general, as well as on their release and fate over the product life cycle. This is acknowledged as a limitation of this study, as additive production can account for a non-negligible portion of 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 (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 plastics routed to biological treatments or subject to in-situ degradation).

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requirement of ensuring similar shelf lives incorporated in the functional unit implies that bottles with similar permeability properties would be used.

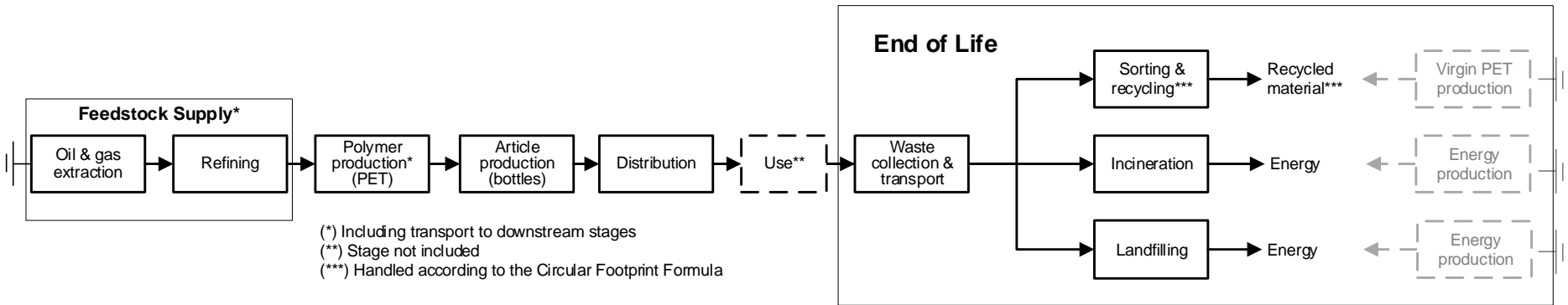


Figure 4.1. System boundary for fossil-based PET beverage bottles (Scenario 1).

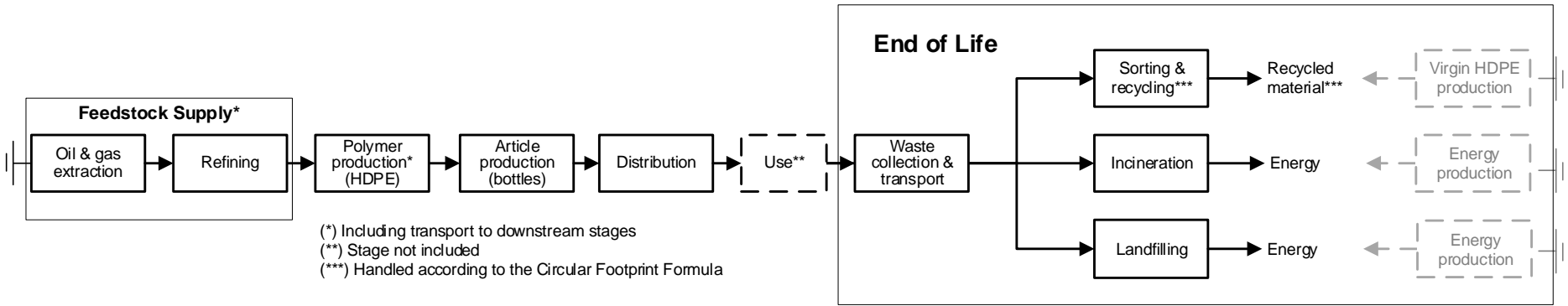
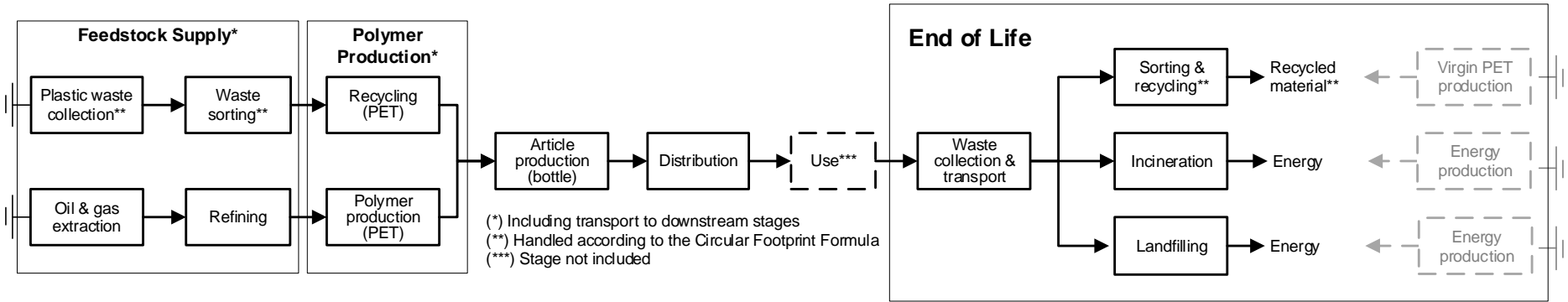
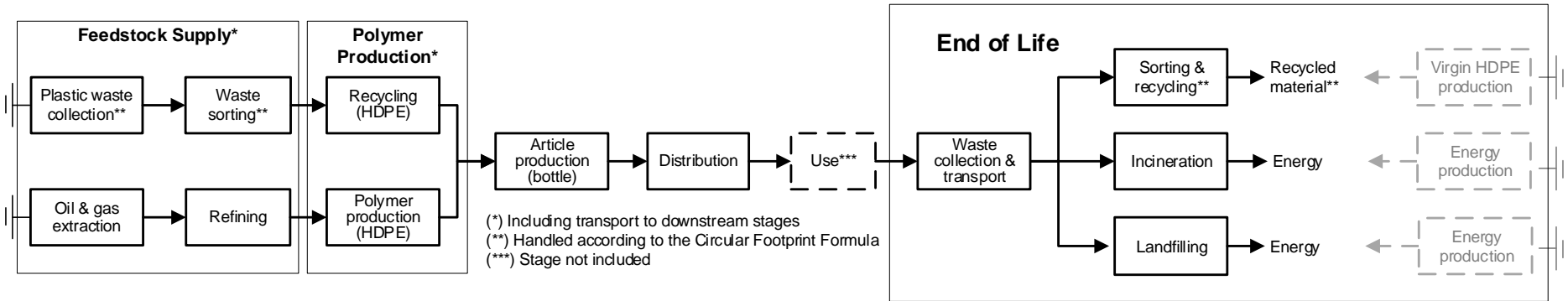


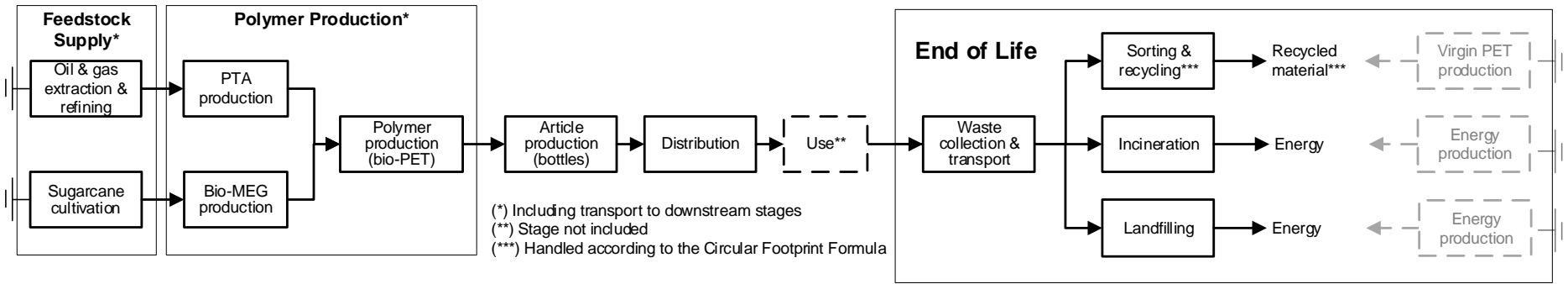
Figure 4.2. System boundary for fossil-based HDPE beverage bottles (Scenario 2).



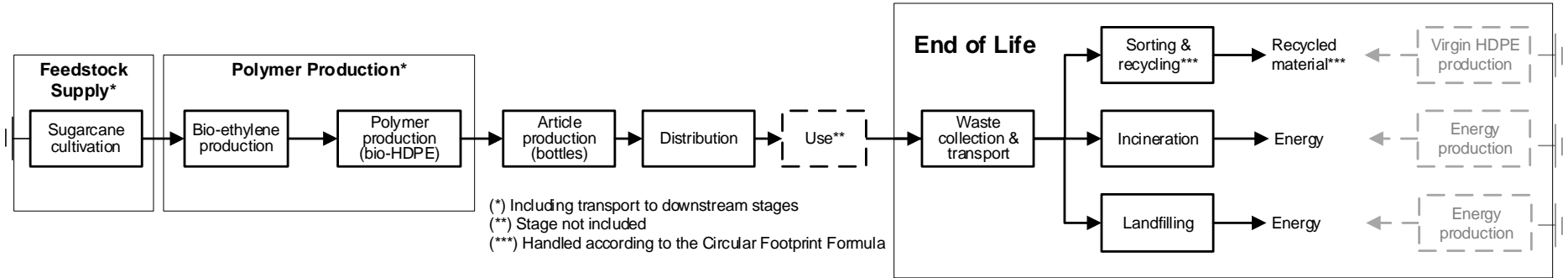
**Figure 4.3.** System boundary for 24% recycled PET beverage bottles (Scenario 3).



**Figure 4.4.** System boundary for 16% recycled HDPE beverage bottles (Scenario 4).



**Figure 4.5.** System boundary for partially bio-based PET beverage bottles (Scenario 5).



**Figure 4.6.** System boundary for bio-based HDPE beverage bottles (Scenario 6).

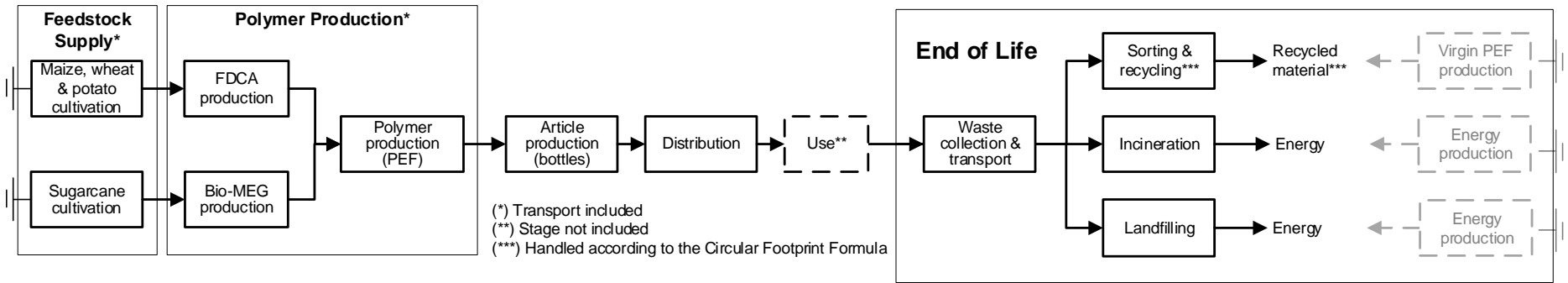


Figure 4.7. System boundary for PEF beverage bottles (Scenario 7).

## 4.4 Life Cycle Inventory

This section describes the overall approach for developing the Life Cycle Inventory of the analysed scenarios, along with the related assumptions and data sources. The description is separated by major lifecycle stages, in the following sub-sections (4.4.1 – 4.4.6). The list of processes, related data sources, and main modelling details are provided in Tables B.1.1 to B.1.7 in Annex B.1.

### 4.4.1 Feedstock Supply Stage

#### 4.4.1.1 Fossil-based polymers

For fossil-based polymers (PET and HDPE), the stage of Feedstock Supply includes the processes of crude-oil and natural gas extraction and transport, naphtha production in crude oil refineries, and its subsequent transport to downstream conversion processes (typically 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. 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 have taken 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 inventory, including exploration activities, well drilling, crude oil extraction and processing, long-distance transport via pipeline and (when required) tanker vessels, as well as regional distribution to the final consumer via pipeline. Oil losses occurring during transportation via pipelines or vessels are also taken into account in the datasets, despite LCA typically looks at normal production conditions, disregarding accidents such as those related to oil spills and fires. Handling of oil losses is not explicitly reported, but they are likely accounted in terms of increased oil requirement 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 anyway quite limited, equalling 0.0115 g per kg of crude oil in the case of seawater, and 0.0539 g/kg in the case of freshwater<sup>14</sup>. Oil fires are not mentioned, and likely not accounted. 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 goods, including infrastructure, are not included, according to the applied 95% cut-off rule, based on material or energy flows, or the level of environmental significance.

Similarly to crude oil, an aggregated dataset from the EF database was applied also to the modelling of natural gas supply: "[EU-27] Natural gas mix; technology mix | consumption mix, to consumer | medium pressure level (< 1 bar)". The EU average natural gas supply mix is represented in the dataset, covering both domestic production

<sup>14</sup> Note that ecotoxicity impacts of these emissions are not assessed, due to the absence of the corresponding characterisation factors in the 2.0 version of the Environmental Footprint LCIA method applied in this project. Characterisation factors for unspecified oil emissions are available in the most recent (3.0) version of the method and can be applied in future evaluations based on this method. Note, however, that characterisation factors for oil (calculated as averages of factors for several refinery products) are lower compared to those of a large portion of substances covered in the LCIA method (i.e. 67% of them).

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. Natural gas losses occurring during transport are also accounted for 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 applied 95% cut-off rule, based on material or energy flows, or the level of environmental significance.

Naphtha production was modelled based on an aggregated gate-to-gate dataset provided by Thinkstep. The dataset represents a (mass-weighted) average refining process for Europe in terms of refining technologies and product outputs, and is based on a dedicated oil refinery model. This is built by largely relying on statistical data and measurements from more than one-hundred (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 input or output. The crude oil demand of a specific unit process is allocated to the respective products and/or intermediate products based on energy (i.e. net calorific value of the product), assigning larger shares of upstream supply burdens to product with higher calorific values. Energy inputs (thermal energy, steam and external electricity) are allocated based on the mass share of the product or intermediate product, out of the total mass of products from the same unit process. With this approach, products requiring more processing steps for production are assigned higher energy consumption burdens. Direct emissions to the environment are allocated based on mass, as well.

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

#### **4.4.1.2 Recycled polymers**

For recycled polymers (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. These processes were modelled as described in Section 4.4.5 (addressing End of Life modelling), and refer to collection and transport of separately collected plastic waste at the municipal level (where most of the feedstock for recycled polymers used in bottle manufacturing normally comes from), as well as to sorting of mixed plastic waste in dedicated facilities. Both processes were implemented in the lifecycle model according to the Circular Footprint Formula (CFF), which is the default approach to handle recycling situations in the Product Environmental Footprint (PEF) context. According to this approach, only 50% of the burdens from collection, transport, sorting and recycling were assigned to the recycled content in beverage bottles ( $A=0.5$ ).



for PET used in bottles, PE used in unspecified applications and for generic plastic packaging), the rest being assigned to the system providing material for recycling. Further details on the implementation of the CFF are provided in Section 4.4.2.2 on recycled polymers production.

#### **4.4.1.3 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.

Growing of Brazilian sugarcane (used as a feedstock for bio-MEG and bio-Ethylene) was modelled through aggregated datasets from the GaBi database. Used in combination, these datasets depict a situation where 45% of sugarcane is manually harvested via the "slash and burn" practice, i.e. sugarcane residues (tops and leaves) are burned on standing plants before harvesting. This share was considered representative of the current situation. However, the slash and burn 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 phasing out of this practice (see Section 4.7.7.2). Transport of harvested sugarcane to further processing in sugarcane mills for Ethanol 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, wheat, potatoes) was modelled through available EF-compliant datasets. The latter are based on the approach used to develop agricultural inventories in the Agri-footprint database, relying on 5-year average yield data from FAOSTat (2010-2014). Included activities are seeding and seed production, fertilizer and pesticide production and application, capital goods, as well as energy use and transport for field management practices. Allocation of cultivation burdens between 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).

### **4.4.2 Polymer Production Stage**

The Polymer Production stage covers the activities of feedstock processing into relevant intermediates and monomer(s), the polymerisation 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 (4.4.2.1 – 4.4.2.4) describe how these activities have been modelled in the present case study.

#### **4.4.2.1 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 aggregated, gate-to-gate datasets provided by Thinkstep. Inputs includes combinations of crude oil, natural gas and naphtha, depending on the polymer. All conversion processes are assumed to take place in Europe, so that the datasets reflect EU average background conditions in terms of e.g. energy generation, material supply and transport. For both polymers, the main conversion process involved in the supply chain is steam cracking of naphtha and natural gas, delivering the monomer Ethylene, along with Propylene, Butadiene and other relevant intermediate flows (e.g. pyrolysis gas including mixtures of Benzene, Toluene and Xylenes). Other relevant conversion processes are catalytic reforming of naphtha and steam reforming of natural gas. In catalytic reforming, naphtha is processed to produce Benzene, Toluene and Xylenes, with the latter (para-Xylene) being an intermediate in the production of Purified Terephthalic Acid (PTA, a co-monomer of PET). Steam reforming of

1 natural gas generates synthesis gas consisting of Carbon Monoxide and Hydrogen, both  
2 used in the production of Methanol (a precursor of Acetic Acid used as a solvent in PTA  
3 production).

4 As for the steam cracking process of naphtha, the applied inventory is mainly based on  
5 industry data, completed, where necessary, by literature data. Allocation among the  
6 different co-products (Ethylene, Propylene, Butadiene, refinery gas, pyrolysis gas and  
7 Hydrogen) is based on energy, considering the net calorific value of co-products. For  
8 catalytic reforming of naphtha and the resulting reformat output, the same allocation  
9 rules and data sources considered for refinery operations are applied (see Section  
10 4.4.1.1), as the process typically takes place at refining facilities. In the subsequent  
11 separation process of reformat gas into its components (including para-Xylene used for  
12 PTA production), allocation based on energy content (net calorific value) is applied,  
13 instead. The same criteria is applied as well to Carbon Monoxide and Hydrogen separated  
14 (via cryogenic separation) from synthesis gas generate through natural gas reforming.  
15 The final polymerisation process of PET and HDPE is mainly modelled based on industry  
16 data from internationally adopted production processes, integrated with literature data  
17 when needed. No allocation is performed at this stage, being PET and HDPE the only  
18 outputs of the respective production process.

#### 19 **4.4.2.2 Recycled polymers**

20 The production of recycled, bottle-grade PET granulate out of sorted, post-consumer PET  
21 waste was modelled based on the ecoinvent dataset "[CH] polyethylene terephthalate  
22 production, granulate, bottle grade, recycled". The dataset has been developed based on  
23 data from two Swiss recycling facilities, and represents the burdens associated with the  
24 processing of sorted, pure-coloured waste PET bales into bottle-grade PET flakes, through  
25 a number of steps. These include bale opening, metal separation, shredding, air  
26 separation of light-weighting label residues (sent to incineration), flotation (separating  
27 HDPE cap fragments from PET flakes), and a further step where PET flakes are treated  
28 with a Sodium Hydroxide solution and heated to nearly 200°C for decontamination  
29 purposes. Purified PET flakes are finally washed with water and then dried. The inventory  
30 was assumed to include also the Solid State Polymerisation (SSP) process, required to  
31 increase the intrinsic viscosity of recycled PET flakes to a level comparable with primary  
32 PET, although is not totally clear from the dataset documentation whether this step is  
33 actually considered. In the implementation in the model, the original dataset was  
34 adjusted from Swiss to EU background conditions, and background datasets related to  
35 energy supply were replaced with EF datasets. The small amount of recovered HDPE  
36 (0.122 kg/kg R-PET) was assumed to directly replace virgin HDPE (1:1 substitution)  
37 according to the hierarchy for the handling of multifunctionality adopted in the PEF  
38 context and outlined in the Method<sup>15</sup>. Finally, a few adjustments had to be performed to  
39 improve reliability of LCIA results for the Ozone Depletion impact category<sup>16</sup>.

40 The same dataset described above was also applied to approximate the burdens  
41 associated with the production of recycled, bottle-grade HDPE, in the absence of more  
42 specific data for such a process (only data for generic recycled HDPE granulate are  
43 available). The same adjustments reported above for recycled bottle-grade PET were  
44 performed, along with the replacement of the input flow of sorted PET waste with that of  
45 sorted HDPE waste.

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<sup>15</sup> In principle, the Circular Footprint Formula (CFF) should be applied also to this minor recovered waste stream (as described further in this section for recycled PET). However, for the purpose of simplicity, direct substitution was applied, with no relevant effects on the results.

<sup>16</sup> 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 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] Market for chemical, organic), i.e. Acetic Acid, Methanol, Urea, Vinyl Acetate, Ethylene Dichloride, and Formaldehyde. The respective shares were then equally subdivided among remaining chemicals.

According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the recycling process<sup>17</sup> were allocated to the recycled content in PET and HDPE bottles ( $A = 0.5$  for PET used for bottles, HDPE used in unspecified applications and for generic plastic packaging). However, the recycled content carries 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 life cycle providing the recycled material). Since the  $Q_s/Q_p$  factor is equal to 1 for (bottle-grade) PET granules from the SSP process (being their quality comparable to that of virgin granules) the allocated share of virgin PET production burdens is equal to 50% ( $A \times Q_s/Q_p = 0.5 \times 1 = 0.5$ ). The same value was applied to recycled HDPE, being its quality necessarily suitable for use in bottle-grade applications, and hence similar to that of replaced virgin HDPE. Virgin polymer production burdens were modelled as described above (Section 4.4.2.1) for conventional fossil-based PET and HDPE, and in Section 4.4.1.1 for the respective Feedstock Supply.

#### **4.4.2.3 Bio-based polymers**

##### **4.4.2.3.1 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 dataset provided by Thinkstep. The dataset covers the steps of sugarcane processing to 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 are accounted as well in the dataset, similarly to fossil-based PET (Section 4.4.2.1). 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 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.

##### **4.4.2.3.2 Bio-based HDPE**

For bio-based HDPE production, no datasets are available from commercial databases or the set of EF-compliant datasets. All conversion processes 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 considers Brazil as a relevant geography. 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 bioethanol-based polymers investigated in this project (i.e. Bio-PET, Bio-LDPE and Bio-PP), and to better align to the hierarchy for the handling of multifunctionality adopted in the present 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

<sup>17</sup> Including both mechanical recycling and the subsequent solid state polymerisation process in the case of PET.

had to be performed, to improve reliability of LCIA results for the impact categories of Ozone Depletion and Resource Use – minerals and metals<sup>18</sup>. The specific amount of sugarcane required for Ethanol production is equal to 15 kg per kg of Ethanol.

Ethanol produced in Brazil was assumed to be transported to Europe for further conversion and polymerisation, consistently with the assumption performed in the aggregated, gate-to-gate datasets used to model the production of the other bioethanol-based polymers considered in this project. Transport is modelled according to the default transport scenario specified in the PEF context (and in the present 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 Ethanol 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.com<sup>19</sup>, 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 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 producers of bio-based HDPE is located in Brazil. In this specific case, the whole process chain of conversion and polymerisation would take place in Brazil, with the final HDPE resin being eventually transported to Europe. The assumption performed 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), and hence a higher mass of material is transported per functional unit compared to the transport of the final HDPE resin.

The inventory for 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 report were combined with background EF datasets for energy generation, and ecoinvent background datasets for material production<sup>20</sup>. Reported transport activities were not implemented in the inventory, being transport of Ethanol (the main raw material) accounted for separately in the foreground inventory, while transport of the other input materials is included in the datasets applied to model their supply. Beyond Ethylene, a small amount of naphtha is also obtained as a co-product, which is handled via direct substitution of naphtha from crude oil refinery (modelled as described for the “Feedstock Supply” stage in Section 4.4.1.1). The data source does not report the specific Ethanol requirement, which was thus determined as the average of the values reported in IEA-ETSAP & IRENA (2013; 1.74 kg Ethanol/kg Ethylene) and IfBB (2018; 2.08 kg Ethanol/kg Ethylene), corresponding to an estimated consumption of 1.91 kg Ethanol per kg of Ethylene.

The final polymerisation step of Ethylene to HDPE was modelled based on data from the most recent PlasticsEurope ecoprofile (PlasticsEurope, 2016a), as implemented in the ecoinvent database. Indeed, inventory data limited to the polymerisation stage 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 this process across all the investigated HDPE-based scenarios (i.e. fossil, recycled, and bio-based HDPE bottles). The applied data represent average values of data collected from

<sup>18</sup> Infrastructure processes related to the Ethanol 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 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.

<sup>19</sup> Available at: <https://www.searates.com/services/distances-time/>

<sup>20</sup> 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.

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 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 production<sup>21</sup>. No allocation or 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 (2017), i.e. 1.07 kg Ethylene/kg, representative of suspension polymerisation only.

#### 4.4.2.3.3 Polyethylene Furanoate (PEF)

Similarly to bio-based HDPE, no life cycle inventories production are available for PEF production in commercial databases or in the set of EF-compliant datasets. The process chain to convert the different feedstock sources into the final polymer was hence modelled based on a combination of existing datasets, 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 dataset provided by Thinkstep, consistently with the dataset used to model partially bio-based PET production from sugarcane. The dataset covers the steps of sugarcane processing to 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 PEF context (and the present 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, wheat, potatoes) 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. In the short term, HMF production from fructose was suggested as a technically and economically preferable alternative, thanks to 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, and was

<sup>21</sup> 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 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] Market for chemical, organic), i.e. Acetic Acid, Methanol, Urea, Vinyl Acetate, Ethylene Dichloride, and Formaldehyde). The respective shares were then equally subdivided among remaining chemicals.

thus proposed as a possible upgrading option for fructose-based plants once production has become more established. In this study, the glucose route was considered, due to the lack of data on fructose production from high fructose syrups, and of suitable (e.g. sufficiently disaggregated) data for the production of the latter out of starch. Starch and glucose production were assumed to take place in the same facility, and no transport was thus modelled between these stages. The same assumption was applied to the conversion of glucose into HMF and to its final polymerisation with bio-MEG into PEF.

Starch production via wet milling of the different starch crops was modelled based on life cycle inventory data reported for the corresponding processes in the Agri-footprint database (v 4.0), combined with background EF-compliant datasets for individual inputs and outputs. In these inventories, the allocation of process burdens to the different co-products (e.g. from wet milling) is based on the respective economic value, consistently with the fully vertically aggregated EF datasets available for starch production via wet milling (which are developed by the same data provider). Maize starch 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.

The inventory of glucose production was based on the ecoinvent dataset "[RER] Glucose production", which refers to pure glucose produced via enzymatic hydrolysis of dried (maize) starch. In the implementation 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 (e.g. for steam production). In addition, the default input of maize starch was replaced with the EU-average mix of starches from European starch crops considered in this case study (i.e. 47% maize starch, 40% wheat starch, and 13% potato starch; see Section 4.1). Note that the inventories applied to model the starch input (described in the paragraph above) refers to the production of dried starch (i.e. starch with a low water content), although starch slurry seems to be actually used by industries for further conversion to glucose and glucose syrups (Starch Europe, 2019). However, dried starch (@14% water content) is considered in the applied ecoinvent dataset for glucose production, and datasets covering the sole production of starch slurry are only available for maize starch. Therefore the original dried maize starch input 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).

Glucose was assumed to be transported to a separate facility where HMF is produced and subsequently polymerised to PEF. This transport was modelled according to the default transport scenario specified 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 inventories are available in the existing databases or in the set of EF-compliant datasets. A new inventory was thus developed based on the results of the process simulations conducted by Motagamwala et al. (2019). The resulting input and output flows were then combined with background EF-compliant datasets to model the respective burdens. Simulation results 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 year. As discussed above, the glucose-based route was chosen, where glucose is firstly isomerised to fructose, and then dehydrated to HMF. To develop the final inventory (summarised in Table 4.4), data reported in the mentioned source were complemented with additional information and data acquired via personal communications with the authors. As already discussed in Section 4.1, the developed inventory is not representative of full-scale industrial production, and do not account for any additional (efficiency) improvement that may take place thanks to further scale-up and process optimisation. However, the underlying simulation data already take into

account the implementation of possible process optimisation strategies (i.e. energy integration).

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 PTA 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). 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 CO<sub>2</sub> (Eerhart et al., 2012). In addition, 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 process optimisation in the PTA production process, the latter is 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 or 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 datasets for energy generation and ecoinvent background datasets for material production.

A similar approximation was also applied to the final polymerisation step of FDCA and bio-MEG to produce PEF, which was assimilated to PET polymerisation out of fossil-based PTA and MEG. Polymerisation of PEF co-monomers operates at lower temperatures than polymerisation of PET (nearly 30°C less), and twice as fast (Eerhart et al., 2012). However, for the same reasons reported above for FDCA production (i.e. the moderate temperature differential and the many decades of process optimisation in PET polymerisation), this process is considered a reasonable (albeit potentially conservative) 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 substitutes (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 datasets for energy generation and ecoinvent background datasets for material production.

1 **Table 4.4.** Life cycle inventory applied for the production of 1 kg of Hydroxymethylfurfural at 99% purity (HMF; based on Motagamwala et al., 2019).

Process section	Stream	Dataset / Flow	Source	Quantity	Unit	Comments
<b>Inputs</b>						
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 {53aeefbc-6896-4630-a654-fb3a5bc559a7}	Plastics Europe (TS)	0.0313	kg	Share of different processes derived from the <i>ecoinvent</i> dataset for acetone supply to the market "[RER] Market for acetone, liquid"
		[RER] Acetone from isopropanol production, technology mix   production mix, at plant   100% active substance {6a377455-759c-4a39-a18f-6a0d58f14853}	EF	0.00704	kg	
		[RER] Acetone from cumene production, technology mix   production mix, at plant   100% active substance {f6d691ae-025e-4c32-bee2-6137c7bbc60a}	EF	0.00211	kg	
HMF purification	MIBK (1:1 on a mass basis)	[GLO] Methyl isobutyl ketone (4-methylpentan-2-one), Technology mix   Production mix, at plant   {c3a1b77e-5c37-4b36-8604-752d4fd79135}	EF	0.0518	kg	-
Energy	Electricity	[EU-28+3] Electricity grid mix 1kV-60kV AC, technology mix   consumption mix, at consumer   1kV - 60kV {34960d4d-af62-43a0-aa76-adc5fcf57246}	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 <sup>(1)</sup>	MJ	Based on the <i>ecoinvent</i> dataset "[RoW] Cooling energy, from natural gas, at cogen unit with absorption chiller 100kW", adjusted to reflect average EU 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 <sup>(1)</sup>	MJ	



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Other inputs	Activated carbon	[RER] activated carbon production, granular from hard coal {3a195a7b-bb9a-4c17-8dcd-bbce8aacc9e6}	EI	1.56E-04	kg	Lifespan: 3 months, after which it is sent to regeneration
	Sn-Beta catalyst	[GLO] Zeolite, from aluminium hydrate, sodium silicate and sodium hydroxide   single route, at plant   2- 2.5 g/cm3 {6e552512-9671-4ea2-a908-a3f42e1f4b0c}	EF	4.79E-03	kg	Sn-containing (Beta) zeolite Approximated with zeolite 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 {85cf81b2-88ed-4f20-ae11-b4a45e356924}	EF	4.22E-04	kg	Benzenesulfonic acid, ethenyl-, polymer with diethenylbenzene Approximated with Naphthalene Sulfonic Acid The total amount is assumed to be replaced every 3 months and then disposed of
<b>Outputs</b>						
Acetone/H <sub>2</sub> O recovery	Excessive H <sub>2</sub> O (H <sub>2</sub> 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 {db00901d-338f-11dd-bd11-0800200c9a66}	TS	0.449	kg	Process-specific burdens for wastewater treatment. Emissions were modelled separately as reported below
	<i>H<sub>2</sub>O to water (91%)</i>	<i>Processed water to river</i>	TS	0.408	kg	Only processed water
	<i>Acetone to water (9%)</i>	<i>Acetone (dimethylcheton), Organic emissions to fresh water</i>	TS	6.46E-04	kg	Based on BOD removal efficiency (98.4%)
	<i>CO<sub>2</sub> to air</i>	<i>Carbon dioxide - Inorganic emissions to air</i>	TS	0.0904	kg	CO <sub>2</sub> from biodegradation of acetone (based on 98.4% removal efficiency)
	<i>H<sub>2</sub>O to water</i>	<i>Water - Other emissions to fresh water</i>	EI	0.0370	kg	H <sub>2</sub> O from biodegradation of acetone (based on 98.4% removal efficiency)
	Humins (to disposal)	<a href="#">[EU-28] Treatment of humins, municipal incineration</a> Developed based on the Doka (2009a) tool for the modelling of material incineration in MSW incineration plants, and background EF and <i>ecoinvent</i> datasets (for energy and material inputs, respectively)	-	0.0862	kg	Humins composition: 56.7% C; 5.4% H; 37.9% O (Agarwal et al., 2017)
HMF purification	Purge (MIBK, H <sub>2</sub> 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 {db00901d-338f-11dd-bd11-0800200c9a66}	TS	0.0661	kg	Process-specific burdens for wastewater treatment. Emissions were modelled separately as reported below

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	MIBK to water (70.1%)	Methyl isobutyl ketone - Organic emissions to fresh water	TS	7.41E-04	kg	Based on BOD removal efficiency (98.4%)
	CO <sub>2</sub> to air	Carbon dioxide - Inorganic emissions to air	TS	0.120	kg	CO <sub>2</sub> from biodegradation of MIBK (based on 98.4% removal efficiency)
	H <sub>2</sub> O to water	Water - Other emissions to fresh water	EI	0.0492	kg	H <sub>2</sub> O from biodegradation of MIBK (based on 98.4% removal efficiency)
	H <sub>2</sub> O to water (27.8%)	Processed water to river - Other emissions to fresh water	TS	0.0184	kg	Only processed water
	Acetone to water (1.6%)	Acetone (dimethylcheton) - Organic emissions to fresh water	TS	0.0225	kg	Based on BOD removal efficiency (98.4%)
	CO <sub>2</sub> to air	Carbon dioxide - Inorganic emissions to air 2/1/2019 {5858ac4c-d8dd-4d08-b1c2-3b0215f757cb}	TS	2.37E-03	kg	CO <sub>2</sub> from biodegradation of acetone (based on 98.4% removal efficiency)
	H <sub>2</sub> O to water	Water - Other emissions to fresh water {1dd04c88-7207-493f-b819-d95a2f94dbca}	EI	9.69E-04	kg	H <sub>2</sub> O from biodegradation of acetone (based on 98.4% removal efficiency)
	HMF (0.5%)	-	-	3.30E-04	kg	No 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 {6a041017-c8ee-4283-9278-f14fdc8656dd}	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 {3a195a7b-bb9a-4c17-8dcd-bbce8aacc9e6}	EI	1.41E-04		Net replaced activated carbon from regeneration (resulting in a total virgin production of 1.56 E-5 kg)
	Disposal of Sn-beta catalyst	[RoW] treatment of waste zeolite, inert material landfill {4b9dd8e7-ae5d-42ea-9f69-679c6bd8790a}	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 {30ac3ea0-c2b3-4abb-882f-de5256434eaa}	EI	4.22E-04		Avoided electricity and heat generation are additionally included in the dataset: 0.5 MJ/kg electricity (net production) 1.18 MJ/kg heat (net heat)

(<sup>1</sup>) Corresponding to the energy demand for “cooling water” and “refrigeration”. Since these value may appear overestimated, scenario impacts were also calculated by omitting these contributions, and results discussed accordingly. Note, however, that this is an optimistic scenario.

**4.4.2.4 Transport of polymer granulate to the article production site**

Modelling of transport of polymer granulate from the polymerisation plant (inside or outside the EU), to the bottles manufacturing and filling site in Europe, is based on the default transport scenario (distances and vehicle types) specified in the PEF context (and in the 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:

- i) 130 km by lorry (total weight >32 t; Euro 4);
- ii) 240 km by train (average freight); and
- iii) 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 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)<sup>22</sup>. Countries contributing to at least 90% of the overall import were considered in the calculation, for an overall distance equal to 10,796 km for fossil-based PET, and to 7,982 km for fossil-based HDPE (see Tables 4.5 and 4.6). Oceanic ship transport was complemented with road transport to and from the harbour in both the exporting and importing country (i.e. the EU). Road transport is made by lorry (total weight >32 t; Euro 4) along an overall default distance of 1000 km.

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

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<sup>22</sup> Available at: <https://www.searates.com/services/distances-time/>

**Table 4.5.** Calculation of the overall average sea distance for imports of fossil-based PET to Europe.

Exporting country	Import <sup>(1)</sup> (%)	Import (% cum.)	Distance <sup>(2)</sup> (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>

<sup>(1)</sup> Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

<sup>(2)</sup> 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 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.

**Table 4.6.** Calculation of the overall average sea distance for imports of fossil-based HDPE to Europe.

Exporting country	Import <sup>(1)</sup> (%)	Import (% cum.)	Distance <sup>(2)</sup> (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	-	-
Overall weighted distance				7,982

<sup>(1)</sup> Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

<sup>(2)</sup> 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 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.

#### 4.4.3 Article production 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, in the future, to PEF bottles. As discussed in Section 4.3, 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 were modelled through the aggregated, EF 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%), 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, industrial scraps. For further detail on the modelling of recycling and avoided virgin material production (including the implementation of the Circular Footprint Formula), the reader is hence referred to Section 4.4.5 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 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 conversion efficiency of the process is equal to 99.9%. Similarly to 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-HDPE currently available on the market is negligible, equalling 0.2% only). 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 4.4.5 on End of Life modelling.

#### 4.4.4 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 PEF context (and in the Method) for the route factory → retail → final client. The following routes were thus considered:

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

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

#### 4.4.5 End of Life Stage

This section describes the modelling of the End of Life stage of beverage bottles. In particular, Section 4.4.5.1 focuses on the definition of the EU-average End of Life scenario, which is considered as a base case for the calculation of the potential impacts of the compared LCA scenarios. The remaining sections (4.4.5.2 – 4.4.5.5) address the modelling of waste collection and transport, and of the different End of Life options explored in the study. Finally, Section 4.4.5.6 provides case-specific details on the estimate of the potential contribution of the beverage bottles to macro-plastics formation at the End of Life (including product litter) and micro-plastic generation throughout the supply chain.

##### 4.4.5.1 End of Life scenario

The same EU-average End of Life scenario was considered for PET bottles, regardless of the type of 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 includes 60% mechanical recycling, 21% incineration, and 19% landfilling. The recycling rate was estimated based on the results of the latest Annual Survey on the European PET Recycling Industry (ICIS and Petcore Europe, 2018). According to the latter, 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 recycling 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 recycling rate reported for PET bottles in Annex C to the PEF method (i.e. 49%, if measured at the input of the recycling process)<sup>23</sup>.

No data on the amount of PET bottles incinerated or landfilled are available. However, average incineration and landfilling rates of total plastic packaging waste can be estimated based on statistics on plastic packaging waste management in Europe for the years 2014-2016 (Eurostat, 2019b). Assuming that all the packaging waste sent to recovery operations other than recycling is incinerated, it is possible to estimate an incineration rate equal to 31%. Similarly, assuming that all the generated waste that is not recovered is landfilled, a landfilling rate of 28% can be estimated. In relative terms, 53% of plastic packaging waste that is not recycled is incinerated, while 47% is 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).

A similar approach was followed also for HDPE bottles (from all types of feedstock), leading to an EU-average End of Life scenario including 64% recycling, 19% incineration, and 17% landfilling. The recycling rate was defined based on the results of the analysis of plastic packaging waste flows in Europe for the year 2014 reported in Deloitte (2017), which is based on extrapolations at the EU level of plastic waste flows in Germany, France, UK, Spain and Italy. The analysis estimates a collection rate for 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 leads to an overall collection rate for HDPE bottles equal to 64%, which has been considered as a recycling rate for modelling purposes. 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 shares are equal to 19% for incineration, and 17% for landfilling.

As for PEF bottles, the same End of Life scenario as PET bottles was assumed, in the absence of specific data for this material, which has not yet been introduced into the market. Since PEF is intended as a potential, fully bio-based replacement for PET in bottles production, a similar collection and recycling scheme could be reasonably established, and similar recycling rates may be achieved (provided that sorting and recycling plants are equipped with suitable devices for proper separation of this polymer stream). An identical End of Life scenario also prevents a biased comparison between the two alternatives.

#### **4.4.5.2 Modelling of waste collection and transport**

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 4.7). These data refer to a region with a well-developed waste management scheme in northern Italy, and can be considered representative of several regions in Europe where good levels of separate collection are achieved, with the implementation of kerbside collection systems. While this may not still be a common practice across all Europe, the approximation is considered reasonable, due to the typically moderate contribution of waste collection and transport to the overall End of Life (and lifecycle) impacts (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 the considered collection pathways as described in Table 4.7.

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

<sup>23</sup> The conversion of the “output recycling rate” reported in Annex C (42%), which is measured at the output of the recycling process, was made by considering a recycling efficiency of 85.5% (consistently with the recycling dataset applied for End of Life recycling; Section 4.4.5.3), and assuming a 100% sorting efficiency for the bottles waste stream (according to the modelling assumption performed in this study for the sorting process; Section 4.4.5.3).

1 derived as well from Rigamonti et al. (2013), and implemented in the model as detailed  
2 in Table 4.8. Note that when EF incineration and landfilling datasets were used, no  
3 transport of collected waste was separately modelled, since the respective burdens are  
4 already accounted for in such datasets. In this case, only collection was modelled,  
5 according to the approach described above.

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**Table 4.7.** Modelling of source-separated plastic waste collection and transport for recycling <sup>(1)</sup>.

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

<sup>(1)</sup> Based on Rigamonti et al. (2013).  
<sup>(2)</sup> Approximating delivery by car or small vans by citizens.

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#### 4.4.5.3 Modelling of sorting and recycling

Before recycling, bales of separately collected plastic waste containing post-consumer bottles are firstly 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.

An average life cycle inventory of mixed plastic waste sorting has been developed in Franklin Associates (2018), based on input/output data from different dual-stream and single-stream sorting facilities in the United States. This inventory has been considered as a reference for the modelling of the sorting process, complementing the reported input and output data with background EF datasets representative of EU-average conditions. In the implementation, a 100% sorting efficiency was assumed for the bottle waste stream, which entirely consists of one of the targeted materials for recycling (caps and labels are excluded from the system boundary in this study). Moreover, no burdens from the treatment of any impurities sorted out as rejects were assigned to the bottle waste stream, to avoid falsely “punishing” them with burdens from unrelated product waste (and mostly coming from incorrect citizen behaviour). Therefore, the modelled sorting process only accounted for the material and energy inputs required to carry out this waste treatment activity. The final inventory of the sorting process applied in the modelling is summarised in Table 4.9.

**Table 4.9.** Life cycle inventory of source-separated mixed plastic waste sorting.

Flow	Amount	Unit	Dataset	Database
<b>Inputs</b>				
Unsorted plastic waste	1	kg	- <sup>(1)</sup>	-
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 <sup>-4</sup>	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}	ecoinvent
<b>Outputs</b>				
Sorted plastic waste	1	kg	-	-

<sup>(1)</sup> A 100% sorting efficiency was assumed for the relevant plastic waste article, as discussed in the text. However, the amount of individual process exchanges was calculated taking into account the actual input of unsorted plastic waste they referred to in the original source (Franklin Associates, 2018).

Mechanical Recycling of PET bottles was modelled through an aggregated EF dataset representing the burdens of secondary PET granulate production out of sorted, post-consumer plastic waste via grinding, metal separation, washing, and extrusion to pellets. The dataset, developed based on literature data for these unit operations, accounts for an overall recycling efficiency equal to 85.5% (on the input material), with process waste and scrap being sent to incineration. This assumption is in line with the typical fate of

recycling residues, which due to their high calorific value are normally sent to incineration or co-combustion in cement kilns (Rigamonti et al., 2014).

In the absence of specific data for PEF (bottles) recycling, the same process dataset described above for PET bottles was considered as an approximation. It can indeed be reasonably expected that a similar combination of the same unit operations applied to PET bottles recycling (i.e. grinding or shredding, metal separation, washing/flotation, and granulation) would be applied also to PEF ones.

As for HDPE bottles recycling, no specific dataset is available in the PEF context. Hence, the *ecoinvent* dataset “[Europe without Switzerland] Polyethylene production, high density, granulate, recycled” was used as a basis for modelling purposes, in combination with EF background datasets for energy and material supply under EU-average conditions. Since a most recent and expanded version of the original inventory data source is available (i.e. Franklin Associates, 2018, updating Franklin Associates, 2011), the dataset 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 performed in the PET bottle recycling process and with the typical fate of recycling residues).

Recycled polymer granulate was assumed to replace virgin granulate of the same material, whose primary production burdens were credited to the system. 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)<sup>24</sup>. To account for the lower overall quality of recycled polymers compared to the replaced virgin polymers, a substitution ratio equal to 0.9 was considered for PET and HDPE, according to default values specified in the PEF context for such materials when used in packaging applications. In the absence of more specific indications, the same value was also assumed for recycled PEF replacing virgin PEF. To model the burdens of avoided primary production, the same datasets (or combination of datasets) used for the modelling of upstream production of the relevant polymers were applied (as described in Sections 4.4.2, and 4.4.1 for the related feedstock). This was made for consistency reasons, and to avoid possible distortions by applying different datasets from other sources. In the case of PET, this implies that avoided production of bottle-grade resin was modelled, despite the output from the considered recycling process is not upgraded to such quality. As a consequence the benefits associated with PET recycling are slightly overestimated<sup>25</sup>.

According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the sorting and recycling processes were allocated to the system ( $A = 0.5$  for PET used in bottles and PE used in unspecified applications and for generic plastic packaging). Similarly, only 50% of the benefits from avoided virgin material production were assigned to the system itself.

<sup>24</sup> The share of bio-based PET consumed in Europe was estimated based on the total apparent PET consumption calculated from Procom data for the year 2015 (Eurostat, 2019c; 5323 kt), and the total bio-based PET consumption estimated for the same year in Spekrijse et al. (2019; 214.2 kt) 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 was estimated based on the global production capacity of bio-based PE in 2018 (European Bioplastics, 2019; 200 kt) and that of PE as a whole in 2016 (PlasticsInsight, 2019; 103 Mt).

<sup>25</sup> Note that no datasets for non-bottle-grade PET (amorphous) production are available in the GaBi database. An EF-compliant dataset is available for amorphous PET, but its application provided distorted results (especially in the Resource Use – minerals and metals impact category) and was thus excluded. The application of datasets from other sources (e.g. *ecoinvent*) was not considered, for consistency reasons and to avoid potentially larger distortions.

#### 4.4.5.4 Modelling of incineration

For conventional fossil-based polymers (i.e. PET and HDPE) aggregated, material-specific incineration datasets are available from the EF database, and were applied to model the treatment of bottles made of these materials in a municipal waste incineration plant. Similarly, for bio-based PET and bio-based HDPE bottles, partially aggregated, material-specific inventories from the GaBi database were applied (no EF datasets are available for these polymers).

All the selected datasets are 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 irrespectively of the waste composition, as they are rather considered a function of the concentration in cleaned flue gas that can be achieved thanks to the applied abatement technologies. For these substances (including HCl, HF, NO<sub>x</sub>, VOC, N<sub>2</sub>O, CO, NH<sub>3</sub>, SO<sub>2</sub>, 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 (m<sup>3</sup>/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<sub>x</sub> reduction, affecting reagent consumption for removal of such substance and its final emission with flue gas.

In line with the general approach to handle energy recovery situations adopted in the PEF context and in the Method (Report I), the product system generating the waste material sent to incineration (i.e. the beverage bottles life cycle, in this case) takes the full burdens from the incineration process. However, it is credited for 100% of the benefits from avoided production of conventional energy (electricity and heat) replaced by energy recovered from waste. While in EF incineration datasets these credits are already accounted for in the aggregated inventory, for GaBi datasets they were added to the main process inventory. Therefore, the EU residual electricity grid mix (as modelled in the EF 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 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 includes 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. Also, thermal energy from waste (11%) was excluded, as according to the approach described above for the handling of energy recovery situations, 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 ready-to-use incineration datasets were available. A disaggregated, material-specific inventory was thus developed, based on the most recent version of the calculation tool developed by Doka (2009a) for the modelling of material incineration within municipal solid waste incineration plants. The model operates similarly

1 to the one used for the development of the EF incineration datasets described above,  
2 allowing the practitioner to account for the specific composition and energy content of  
3 the incinerated waste to develop a material-specific incineration inventory based on  
4 transfer coefficients (see Table 4.10 for an overview of the considered PEF composition  
5 and energy content). The tool also allows to adjust other technological parameters to the  
6 relevant geography or scope, including energy efficiencies, the share of alternative NOx  
7 control technologies, and a few other specific parameters. In this case study, energy  
8 efficiencies were adjusted to better reflect the current EU-average situation, while default  
9 values were kept for other parameters, which are representative of modern incineration  
10 plants in central and Western Europe. A gross electricity efficiency equal to 13.7% and a  
11 gross thermal efficiency of 31.8% were estimated for Europe, and applied in the  
12 modelling. These efficiencies account for the share of waste routed to incineration plants  
13 operating without any energy recovery (estimated to be 9% for municipal waste)<sup>26</sup>, while  
14 considering that plants with energy recovery operates with an average gross electricity  
15 efficiency equal to 15.1%, and a gross thermal efficiency equal to 35% (CEWEP, 2012)<sup>27</sup>.  
16 Credits associated with recovered energy were calculated as described above for existing  
17 GaBi datasets (i.e. electricity from the EU-average residual grid mix, and thermal energy  
18 from the current EU-average mix of heat sources). In the final dataset, the inventory  
19 flows generated by applying the Doka (2009a) tool were combined with the background  
20 ecoinvent datasets typically applied within incineration inventories available in such  
21 database. However for energy-related flows, background EF datasets were applied.

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<sup>26</sup> Calculated as the average share of municipal waste incinerated without energy recovery over the years 2015-2017 (Eurostat, 2019d).

<sup>27</sup> 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).

**Table 4.10** Elemental composition and lower heating value of PEF considered for End of Life modelling <sup>(1)</sup>.

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

<sup>(1)</sup> 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 for the modelling of waste PET incineration in the applied EF dataset. 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 allows to perform a consistent modelling for both materials, even without the availability of a full composition analysis for post-consumer PEF in municipal waste.

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

#### 4.4.5.5 Modelling of landfilling

Landfilling of bottles made of conventional non-biodegradable polymers (i.e. fossil-based PET and HDPE, both virgin and recycled) was modelled based on a common, aggregated EF dataset representing disposal of plastic waste in a managed, municipal solid waste landfill (*[EU-28+EFTA] Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific sites)*). The underlying inventory is material-specific, but refers to the average chemical composition and degradability values of generic plastic waste, rather than to those of the specific polymer. 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<sup>28</sup>. 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 emissions to the environment over a 100 year time horizon. Emissions occurring beyond 100 years are not accounted for in the model. Landfill gas generation is calculated based on the organic carbon content in the waste and the respective degradation rate assumed over 100 years (not

<sup>28</sup> While more polymer-specific landfilling datasets are available from other databases (i.e. *ecoinvent*), the mentioned EF dataset for landfilling of generic plastic was selected, as specifically referring to EU as the reference geography (in contrast to available polymer-specific datasets), and to keep consistency with the overall dataset selection “hierarchy” followed in this project (Section 3.5).

reported), while, 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 rate of landfill gas, energy efficiencies of 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 m<sup>2</sup>. 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 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 CO<sub>2</sub> and CH<sub>4</sub> 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 amounts of fossil CO<sub>2</sub> and CH<sub>4</sub> reported as released to air were converted to biogenic emissions (disregarding at the same time the only partial bio-based origin of carbon in Bio-PET bottles). This approximation is considered acceptable, as reported CO<sub>2</sub> and CH<sub>4</sub> emissions are modest, and overall amounting to only 1.4% of the carbon content in the landfilled plastic material, despite they also include the contribution of emissions from background processes. This can be partly explained by the generally low degradation (and mineralisation) of conventional, non-biodegradable polymers in landfill, which is typically assumed to be in the range of 1% over 100 years (Doka, 2009b). On the other hand, it is acknowledged that this approach is in favour of the two bio-based polymers.

In line with the time horizon applied for landfill emission modelling in the selected dataset, (biogenic) carbon in the landfilled polymers that is not degraded (mineralised) after 100 years from deposition was considered to be never released from the landfill body. As a base case, this missed release of atmospheric carbon taken up during biomass growth is not reflected in Climate Change LCIA results for Bio-PET and Bio-HDPE bottles, since characterisation factors for biogenic CO<sub>2</sub> flows have been set to zero (fully conforming to provisions in the PEF method). However, to show the relevance of this choice on the overall results, the Climate Change impact of the two bio-based alternatives was also calculated by acknowledging the effects of non-released biogenic carbon. For this purpose, a specific CO<sub>2</sub> uptake was modelled in the inventory, expressing the net amount of CO<sub>2</sub> taken up during biomass growth and not released (as CO<sub>2</sub> or CH<sub>4</sub>) during polymer degradation in landfill. The uptake was calculated based on the biogenic carbon content in the polymers, and assuming a mineralisation rate over 100 years equal to 1% for both non-biodegradable polymers (Doka, 2009b). While the mineralisation rate assumed in the dataset is not reported explicitly, it seems to be in line with this assumption (according to the figures reported above regarding the inventoried carbon emissions). Considering a biogenic carbon content equal to 12.5% (i.e. 20% of a total carbon content of 62.5%) for Bio-PET and to 81.9% for Bio-HDPE, a carbon uptake equal to 0.454 kg CO<sub>2</sub>/kg Bio-PET and 2.97 kg CO<sub>2</sub>/kg Bio-HDPE was calculated. This uptake was then characterised with a factor of -1 kg CO<sub>2</sub> eq. per kg CO<sub>2</sub>-C not degraded.

For bottles made of non-“drop-in”, bio-based polymers (i.e. PEF), a material-specific landfilling inventory was built, based on the calculation tool developed by Doka (2009b)



for the modelling of waste disposal into sanitary landfills. Similarly to waste incineration, the tool allows to calculate material-specific landfilling inventories accounting for the specific chemical composition and other relevant chemo-physical properties of the landfilled waste (Table 4.10), as well to adjust a number of relevant site- and technology-specific parameters to the reference geography and to the corresponding average landfilling technology. For this purpose, the model applies element-specific transfer coefficients to calculate the distribution of decomposition products originating from elements in the waste composition between landfill gas and leachate, and their ultimate emissions 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. carbon to CH<sub>4</sub> and CO<sub>2</sub>) within the landfill. For non-biodegradable, bio-based polymers (i.e. PEF in this case), the degradability over 100 years from deposition was set to 1%, consistently with the value assumed in the model for conventional, non-biodegradable polymers such as PET and PE. As discussed 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. Also in this case, to calculate a variant of Climate Change LCIA results that acknowledge the effects of non-released atmospheric carbon, a specific CO<sub>2</sub> uptake was modelled in the inventory, expressing the net amount of CO<sub>2</sub> taken up during biomass growth and not released (as CO<sub>2</sub> or CH<sub>4</sub>) during polymer degradation in landfill. Considering a biogenic carbon content in PEF equal to 52% (Table 4.10), a carbon uptake equal to 1.89 kg CO<sub>2</sub>/kg PEF was calculated, based on the assumed 1% mineralisation rate (as reported above).

Site-specific and technology-specific parameters were set so as 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% is flared without any kind of energy recovery (Couturier et al., 2010). Engines were assumed to operate with a net electricity efficiency of 13.5%, and a net heat efficiency of 27.8% (according to the default values assumed in the model). 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).

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 database<sup>29</sup>. However, for energy-related flows, background EF datasets were applied.

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<sup>29</sup> Exceptions are the inputs of diesel (burned in building machine) and pitch, which were replaced with suitable EF datasets, to improve reliability of LCIA results in the Ozone Depletion impact category.

#### 4.4.5.6 Contribution to macro- and micro-plastics generation (including product litter)

The contribution to macro- and micro-plastics generation of the analysed beverage bottles scenarios was estimated according to the framework outlined in Annex B of the Method (Report I). As a base case, a two-level approach was applied to estimate the (total) loss from processes and the release to ocean and to the terrestrial environment of both macro- and micro-plastics throughout the product life cycle. The first level follows the Plastic Leak Project (PLP) method (“PLP method”; Peano et al., 2020); the second level is an expanded version of the latter (“Expanded PLP method”), which accounts for additional contributions (i.e. export of plastic waste outside the EU, road markings and marine coatings) albeit with higher uncertainty. To evaluate the robustness of part of the results obtained by applying such methods, two alternative approaches were also applied as a sensitivity analysis. First, an alternative estimate of the total plastic loss (macro- and micro-plastics) along the supply chain was conducted, by applying the approach developed by Ciroth and Kouame (2019). Moreover, the sole macro-plastics release to ocean at End of Life was alternatively estimated by applying product-specific marine litter rates based on beach litter observations in the EU. In this section, the case-specific parameters considered to apply these methods and approaches to the beverage bottles LCA scenarios are reported.

##### 4.4.5.6.1 PLP method

Regarding the estimate of the loss and release of macro-plastics at the End of Life stage (from product littering and waste mismanagement), the product-specific parameters reported in Table 4.11 were considered to apply the first level of the framework (“PLP method”) to beverage bottles (derived from Peano et al., 2020). Note that such parameters are defined regardless of the type of feedstock or material used for bottles manufacturing. Similarly, all beverage bottles scenarios contribute to macro-plastics loss and release to the same extent, as the chosen feedstock or material do not affect the probability of the article to be littered.

**Table 4.11.** Case-specific parameters of the *PLP method* applied to the beverage bottles LCA scenarios.

Parameter <sup>(1)</sup>	Value
Littering rate ( $LR_{lit}$ ) (%)	2
Release rate to ocean ( $Rel_{ocean}$ ) (%)	10
Release rate to the terrestrial environment ( $Rel_{terenv}$ ) (%)	5

(1) For details on the meaning of each parameter, refer to Annex B of the Method (Report I).

No case-specific parameters are required to calculate the supply-chain loss and release of micro-plastics to ocean and to the terrestrial environment by applying the PLP method. Indeed, the contribution of the considered micro-plastics sources (pellet loss and tire abrasion) depends on the magnitude of the relevant life cycle processes (i.e. polymer production and road transport) in the specific beverage bottles scenario (which in turns depend on the reference flow<sup>30</sup> of each scenario). However, the quantification is made by means of default parameters that are not affected by the type of product, polymer or feedstock source (as reported in Annex B of the Method (Report I)).

##### 4.4.5.6.2 Expanded PLP method

The parameters required to calculate the micro-plastics loss and release from marine coatings and road markings are not case-specific, and are reported in Annex B of the Method (Report I). Similar to the other considered micro-plastics sources, the contribution of these additional sources depends on the intensity of relevant (transport)

<sup>30</sup> The reference flow is the mass of bottle material required to fulfil the functional unit.

processes in the specific product inventory, by means of the above-mentioned case-  
unspecific parameters.

As for the calculation of the additional contribution of plastic waste exported for recycling  
to the loss and release of macro-plastics from waste mismanagement, the case-  
unspecific parameters described in Annex B of the Method (Report I) were applied, in  
combination with the EU-average recycling rates estimated for each type of beverage  
bottles (Section 4.4.5.1). For the sake of clarity, the applied recycling rates are  
summarised again in Table 4.12.

**Table 4.12.** EU-average recycling rates considered for each beverage bottles scenario to estimate  
the amount of waste plastic bottles exported to non-EU countries for recycling.

Polymer	Recycling rate
PET	60%
R-PET	
Bio-PET	
HDPE	64%
R-HDPE	
Bio-HDPE	
PEF	60%

#### 4.4.5.6.3 Sensitivity analysis 1: alternative calculation of the total plastic loss along the supply-chain

The contribution of each (foreground) life cycle process to the total plastic loss of the  
different beverage bottles scenarios was quantified by applying the approach by Ciroth &  
Kouame (2019) to the corresponding ecoinvent dataset or, if needed, adapted dataset  
from the same database<sup>31</sup>. Table 4.13 summarises the datasets considered for the  
cradle-to-gate process-chain involved in the production of each polymer, along with the  
possible adaptations performed. The datasets considered for the remaining foreground or  
background lifecycle processes are instead listed in Annex D. Note that this approach  
excludes the contribution from product littering and waste mismanagement at End of  
Life.

<sup>31</sup> Equivalent to those applied in the actual lifecycle model of each scenario, which could not be directly  
considered for calculation purposes as the mentioned approach has been specifically developed to be  
applied to *ecoinvent* inventories.

**Table 4.13.** Ecoinvent datasets applied and possible adaptations performed to quantify the total plastic loss from the production of the different polymers considered for beverage bottle manufacturing, through the approach developed by Ciroth and Kouame (2019).

Polymer	Dataset	Adaptation
PET	[RER] Polyethylene terephthalate production, granulate, bottle grade	No adaptation required.
R-PETr	Generic recycled plastic dataset developed on purpose.	No adaptation required.
Bio-PET	[RER] Polyethylene terephthalate production, granulate, bottle grade	Fossil-based ethylene glycol (MEG) is replaced with bio-based MEG (approximated with sugar from sugarcane).
HDPE	[RER] Polyethylene production, high density, granulate	No adaptation required.
R-HDPE	Generic recycled plastic dataset developed on purpose.	No adaptation required.
Bio-HDPE	[RER] Polyethylene production, high density, granulate	90% of the original fossil feedstock is replaced with sugarcane.
PEF	[RER] Polyethylene terephthalate production, granulate, bottle grade	Fossil-based MEG is replaced with sugar from sugarcane and PTA is replaced with glucose from starch

#### 4.4.5.6.4 Sensitivity analysis 2: alternative calculation of macro-plastics release to ocean

The EU-average marine litter rate (MLR, %) for beverage bottles and the resulting release to ocean (kg) were estimated based on beach litter observations and apparent consumption data at the EU level, according to the procedure reported in Annex B of the Method (Report I). For calculation purposes, average values for the period 2012-2016 were considered for both quantities. Moreover, the following product categories were considered in the respective data sources, reported as well below:

- Beach litter observations (Beach litter database; Addamo et al., 2018; Hanke et al., 2020): “*Drink/cleaner/cosmetics/food bottles & containers*” (B3), since bottles and containers are aggregated.
- EU Apparent consumption (PRODCOM database; Eurostat, 2019c): “*Plastic carboys, bottles, flasks and similar articles for the conveyance or packing of goods, of a capacity <= 2 litres*”, since no specific category for drink bottles was available.

Table 4.14 summarises the values considered in the calculation, along with the obtained estimates of the beach litter rate and of the resulting (up-scaled) marine litter rate.

**Table 4.14.** Estimate of the beach litter rate and of the resulting up-scaled marine litter rate for beverage bottles.

Total predicted beach litter <sup>(1)</sup> (pieces/year)	Total apparent EU consumption <sup>(2)</sup> (pieces/year)	Beach Litter rate (%)	Marine Litter rate (%)
8.22E+07	1.18E+11	0.071	1.42

<sup>(1)</sup> Along the entire length of EU coast. Estimated based on observed beach litter data over 100 m of coastline (from Addamo et al., 2018 and Hanke et al., 2020).

<sup>(2)</sup> From PRODCOM (Eurostat, 2019c).

#### 4.4.6 Calculation of iLUC impacts

As a base case, the contribution of Indirect Land Use Change (iLUC) to the potential Climate Change impact of bio-based or partially bio-based beverage bottles alternatives was calculated according to the approach outlined in Section 4.4.16.4 of the Method (Report I). A sensitivity analysis on the applied method has been performed, with the respective results being presented in section 4.7.8.3.

In order to apply (recalculated) iLUC GHG emission factors from the EU 2015/1513 Directive (EC, 2015), the specific land demand for the production of relevant crops ( $\text{m}^2 \cdot \text{year} / \text{kg crop}$ ) was calculated first. The calculation was based on the aggregated amount of arable and agriculture land occupation flows reported, for the relevant country (e.g. Brazil for sugarcane), in the dataset used to model the production of the specific crop. If the geography of such flows was not specified, all the arable and agricultural land occupation flows reported in the dataset were aggregated. These estimates were checked against values of land demand calculated based on 5-years average crop yields from FAOSTAT (FAO, 2019), which were found to be generally aligned with the former (absolute variation between 3% and 20%). Hence, the values estimated based on land occupation flows were applied, for consistency with the overall LCI modelling of scenarios.

Specific land demand for crop production was then converted into a demand per functional unit, based on the specific consumption of crop for polymer production ( $\text{kg crop/kg polymer}$ , consistently with the values applied in the LCI modelling) and the amount of polymer needed to fulfil the functional unit (reference flow) in the relevant scenario. The iLUC contribution to the Climate Change impact was finally calculated by applying to the latter the recalculated GHG emission factors from the Directive. The described calculation steps to estimate the iLUC contribution to the potential Climate Change impact are summarised in Table 4.15.

**Table 4.15.** Calculation of the iLUC contribution to the potential Climate Change impact of beverage bottles LCA scenarios relying on bio-based polymers.

Scenario / Polymer	Feedstock	Land demand for crop production <sup>(1)</sup> [m <sup>2</sup> *y/kg <sub>crop</sub> ]	Crop demand for polymer production [kg <sub>crop</sub> /kg <sub>polymer</sub> ]	Polymer demand per F.U. [kg <sub>polymer</sub> /FU]	iLUC GHG emission factor [kg CO <sub>2</sub> eq./ (m <sup>2</sup> *y)]	iLUC climate change impact [kg CO <sub>2</sub> eq./FU]
S5 – Bio-PET	Sugarcane (BR)	0.117 (0.135)	4.27	51.6	0.176	4.53
S6 – Bio-HDPE	Sugarcane (BR)	0.117 (0.135)	28.6	54.3	0.176	32.0
S7 - PEF	Sugarcane (BR)	0.117 (0.135)	4.3	41.3	0.176	3.65
	Maize (EU)	1.34 (1.37)	0.503	41.3	0.0612	1.70
	Wheat (EU)	1.52 (1.75)	0.476	41.3	0.0612	1.83
	Potatoes (EU)	0.376 (0.301)	0.552	41.3	0.0612	0.524
	Total	-	-	-	-	7.71

(1) Based on arable and agriculture land occupation exchanges reported in the dataset applied to the modelling of the specific crop. Values in parenthesis refer to land demand calculated based on crop yield data from FAOSTAT (5-years average), and are reported as a reference.

## 4.5 Life Cycle Impact Assessment results

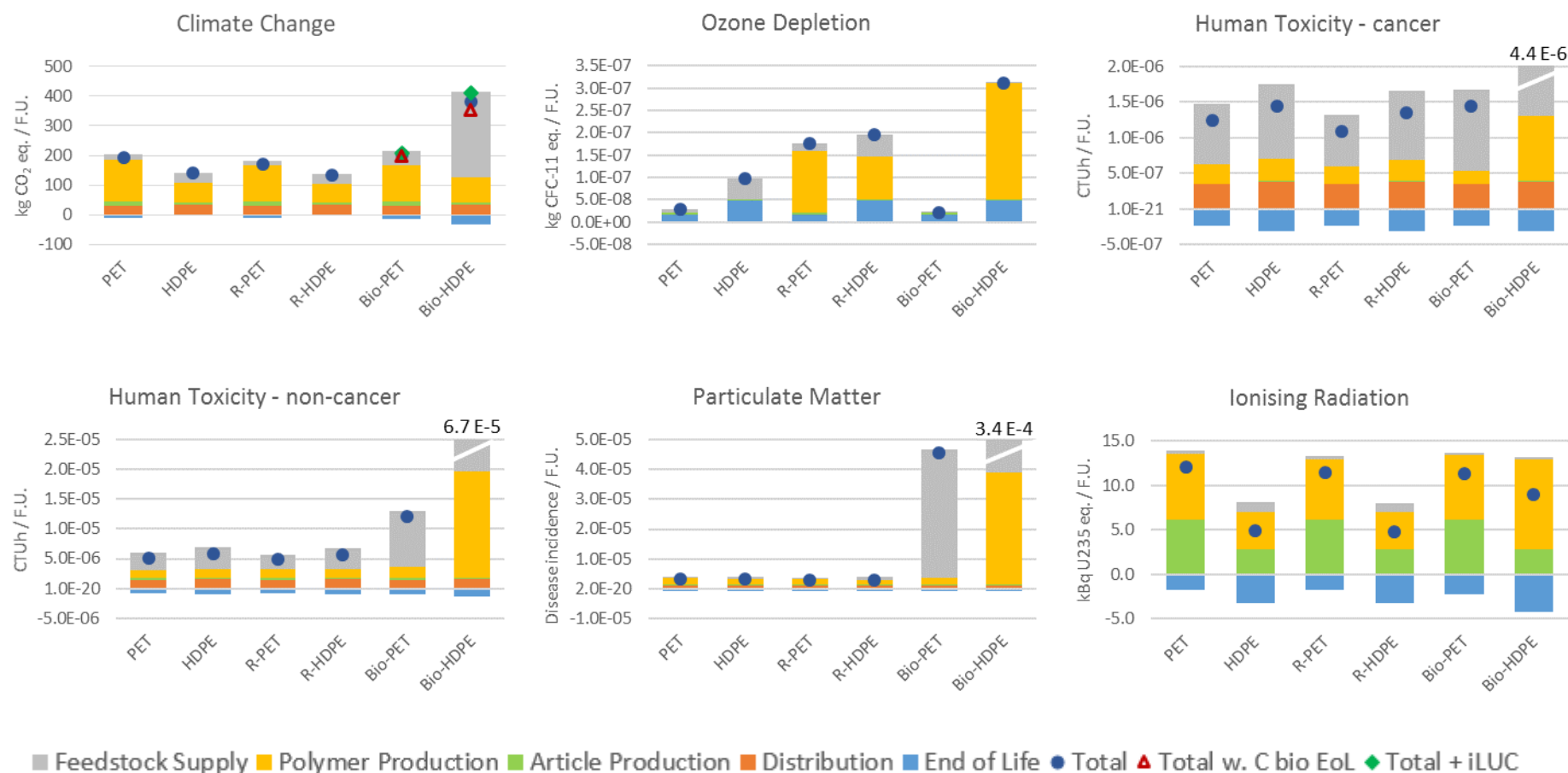
The characterised potential impacts of the examined scenarios (with and without the iLUC contribution to Climate Change) are reported in Figures 4.8 to 4.10. These also show the breakdown of contributions from the main lifecycle stages, which include:

- i) Feedstock Supply, i.e. oil/natural gas extraction, transport and possible refining, transport of naphtha from refinery to downstream users; collection, transport and sorting of post-consumer plastic waste (recycled polymers); or crop cultivation and transport to further processing (bio-based polymers);
- ii) Polymer Production, i.e. all gate-to-gate activities from feedstock processing to monomer/intermediate production and polymerisation, including any transport among these, as well as transport of polymer granulate to the beverage bottles manufacturing and filling site;
- iii) Article Production, i.e. conversion of the polymer into beverage bottles by injection-stretch-blow moulding (PET and PEF bottles) or extrusion-blow moulding (HDPE bottles);
- iv) Distribution, i.e. transport of beverage bottles from the manufacturing and filling site to the final user; and
- v) 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.

Figure 4.11 shows the contribution of each sub-category to the Climate Change impact (i.e. from fossil, biogenic and land use/land transformation GHG emissions), while the overall characterised, normalised and weighted impact assessment results are available in Tables B.1.7.-B.1.9, in Annex B.1. The latter are not reported in this section to keep readability of the document, despite this is not in alignment with the position suggested in the LCA report template provided in the Method (Report I). Results related to PEF bottles are reported separately in Table 4.16, considering the lower level of maturity of some of the processes involved in PEF production, compared to the other examined polymers.

Note that scenario impacts presented in Figures 4.8 to 4.10 refer to the EU-average End of Life scenario (as described in Section 4.4.5.1), and represent the net impacts from waste management activities, resulting from the balance between real burdens and benefits (if any). Potential impacts calculated assuming 100% of post-consumer bottles being routed to each viable End of Life option are presented in Figures 4.12 to 4.14, except for Bio-HDPE and PEF bottles, which for reasons of scale are separately presented in Annex B.1, Tables B.1.10 and B.1.11. We warn that this analysis should not be interpreted as a direct LCA comparison of alternative End of Life management options for beverage bottles, because a product-based perspective is considered, and allocations among consecutive product life cycles are applied to the recycling pathway. Conversely, in a typical waste management LCA that compares alternative End of Life options (i.e. with a functional unit of 1 tonne of waste managed), the recycling pathway would be assigned the full burdens and benefits it involves, as there is no need to break the system mass flows between the system recycling and the one using the recycled feedstock (i.e. system perspective; no allocation needed).

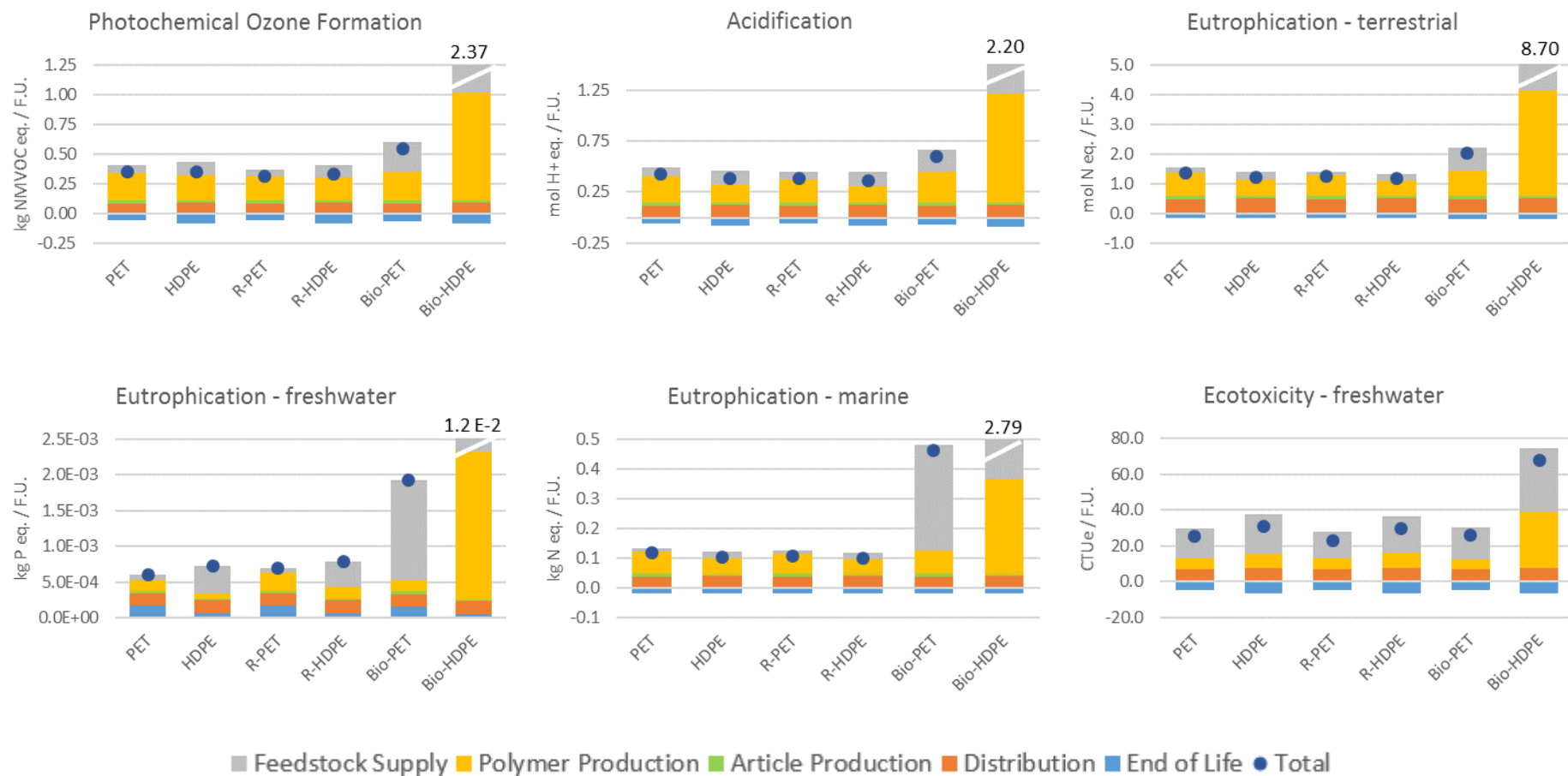
## Comparative LCA of Alternative Feedstock For Plastics Production – DRAFT FOR STAKEHOLDER CONSULTATION –Part II



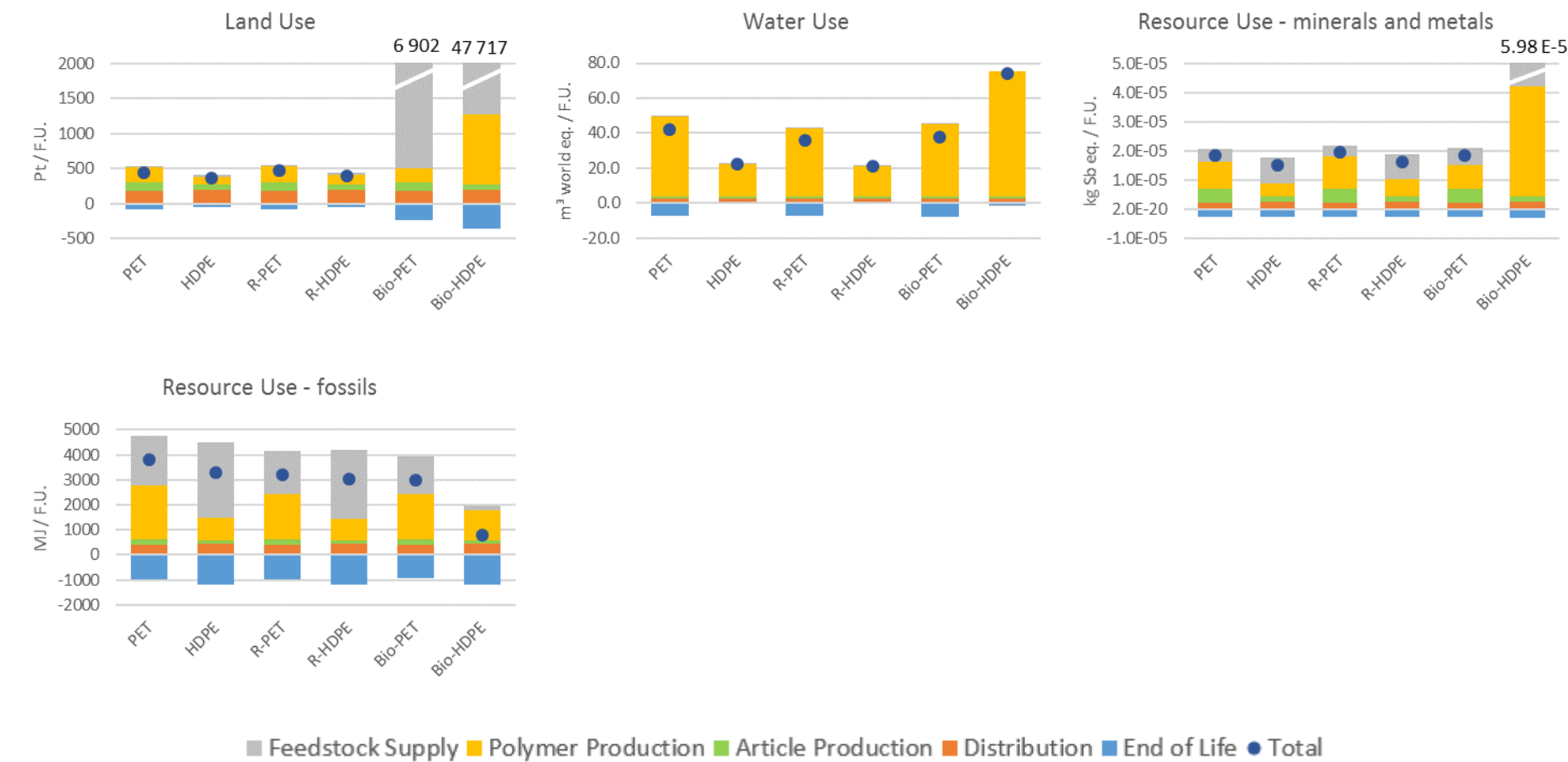
**Figure 4.8.** Potential impact of beverage bottles LCA scenarios for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity - non-cancer, Particulate Matter and Ionising Radiation. Note that in some impact categories a part of the results is out of scale and is curtailed. Climate Change impacts denoted with "C bio EoL" accounts for the contribution of biogenic carbon not released after 100 years from landfilling of bio-based beverage bottles.



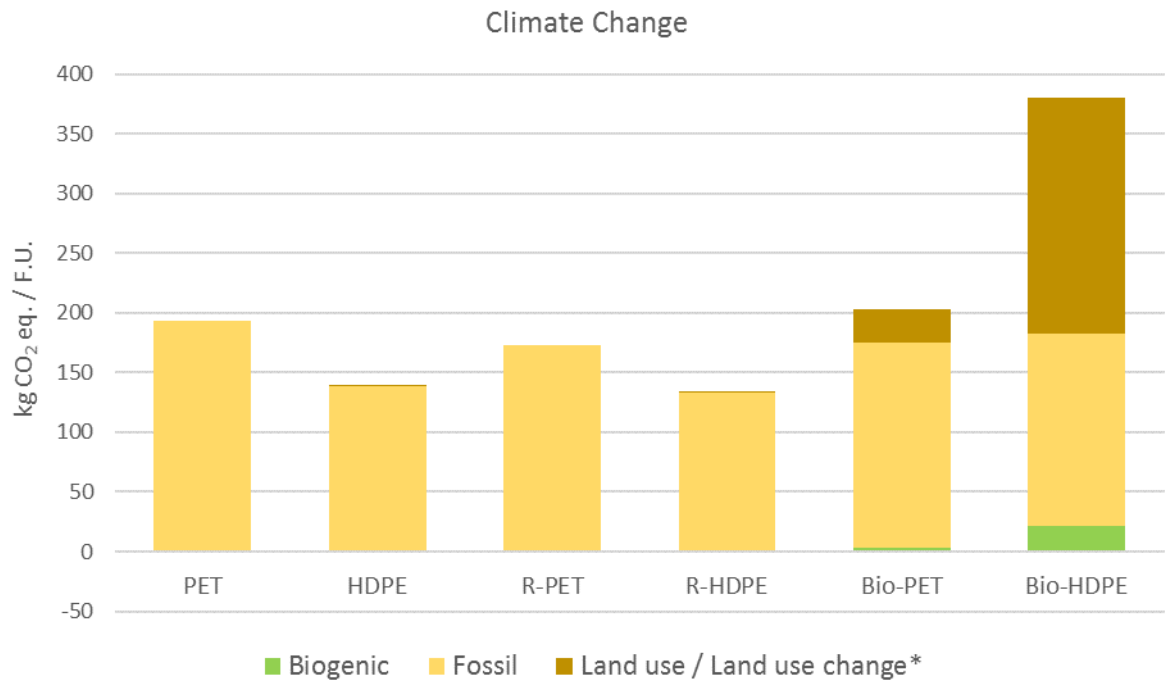
## Comparative LCA of Alternative Feedstock For Plastics Production – DRAFT FOR STAKEHOLDER CONSULTATION –Part II



**Figure 4.9.** Potential impact of beverage bottles LCA scenarios for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater. Note that in some impact categories a part of the results is out of scale and is curtailed.



**Figure 4.10.** Potential impact of beverage bottles LCA scenarios for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils. Note that in some impact categories a part of the results is out of scale and is curtailed.



**Figure 4.11.** Contribution of fossil, biogenic, and land use/land transformation GHG emissions to the total Climate Change impact of beverage bottles LCA scenarios (note that the contribution of iLUC to land transformation emissions is excluded).

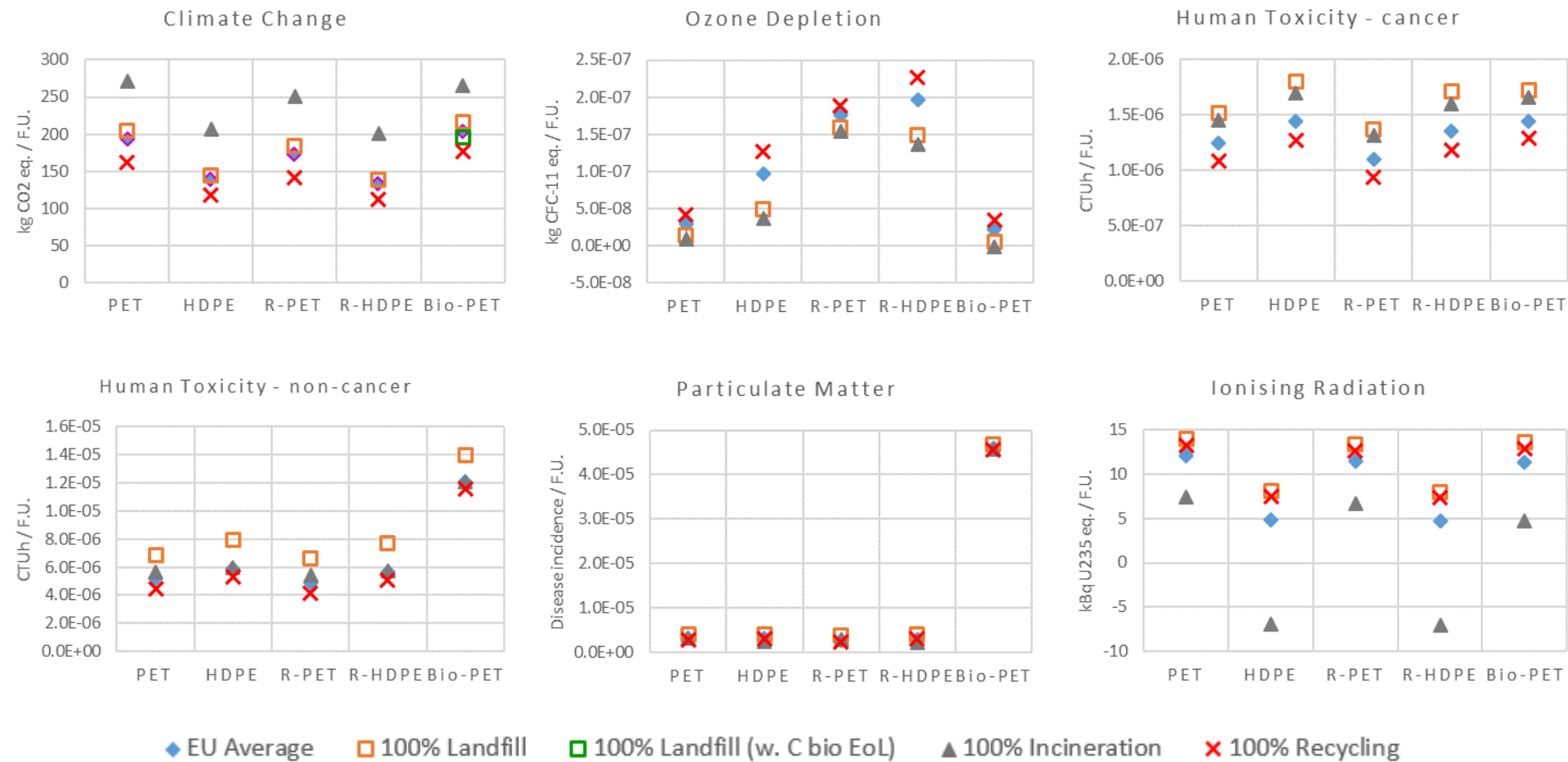
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**Table 4.16.** Potential impacts of PEF beverage bottles (Scenario 7) and contribution of the main life cycle stages.

Impact category	PEF						PET
	Feedstock supply	Polymer production	Article production	Distribution	End of Life	Total	Total
Climate Change [kg CO <sub>2</sub> eq.]	5.57E+01	6.05E+02	2.92E+00	2.55E+01	-1.43E+02	5.46E+02	1.93E+02
<i>Climate Change (w. C bio EoL) [kg CO<sub>2</sub> eq.](<sup>1</sup>)</i>						5.31E+02	-
<i>Climate Change (+ iLUC) [kg CO<sub>2</sub> eq.]</i>						5.54E+02	-
<i>Climate Change (biogenic) [kg CO<sub>2</sub> eq.]</i>						2.82E+00	1.53E-01
<i>Climate Change (fossil) [kg CO<sub>2</sub> eq.]</i>						5.25E+02	1.93E+02
<i>Climate Change (land use change) [kg CO<sub>2</sub> eq.]</i>						1.78E+01	-6.26E-02
Ozone Depletion [kg CFC-11 eq.]	1.55E-07	4.42E-06	-6.36E-08	6.08E-11	-9.89E-07	3.52E-06	2.95E-08
Human Toxicity - cancer [CTUh]	1.56E-06	2.29E-06	-4.42E-08	2.84E-07	-7.73E-07	3.32E-06	1.24E-06
Human toxicity - non-cancer [CTUh]	6.87E-05	2.63E-05	-1.14E-06	1.17E-06	-1.78E-05	7.72E-05	5.18E-06
Particulate matter [Disease incidence]	3.82E-05	9.97E-06	-3.63E-07	6.86E-07	-1.07E-05	3.78E-05	3.09E-06
Ionising Radiation [kBq U235 eq.]	9.97E-01	3.13E+01	4.43E+00	5.37E-02	-7.00E+00	2.98E+01	1.21E+01
Photochemical Ozone Formation [kg NMVOC eq.]	2.32E-01	8.99E-01	6.52E-03	7.20E-02	-2.49E-01	9.61E-01	3.58E-01
Acidification [mol of H+ eq.]	5.97E-01	1.19E+00	6.98E-03	8.83E-02	-3.98E-01	1.48E+00	4.31E-01
Eutrophication - terrestrial [mol N eq.]	2.63E+00	2.97E+00	9.01E-03	5.37E-02	-1.23E+00	4.43E+00	1.39E+00
Eutrophication - freshwater [kg P eq.]	7.56E-03	4.72E-02	-7.80E-04	1.35E-04	-2.03E+02	-2.03E+02	6.04E-04
Eutrophication - marine [kg N eq.]	6.62E-01	4.04E-01	-7.21E-03	3.10E-02	-2.33E-01	8.57E-01	1.19E-01
Ecotoxicity - freshwater [CTUe]	6.28E+02	3.24E+02	-1.36E+01	5.44E+00	-2.03E+02	7.41E+02	2.52E+01
Land Use [Pt]	1.34E+04	1.16E+03	-1.09E+02	1.45E+02	-3.31E+03	1.13E+04	4.47E+02
Water Use [m <sup>3</sup> world equiv.]	1.14E+02	1.52E+02	-2.43E+00	2.19E+00	-5.84E+01	2.07E+02	4.23E+01
Resource Use - mineral and metals [kg Sb eq.]	9.70E-06	1.15E-03	-1.33E-05	1.86E-06	-2.58E-04	8.90E-04	1.84E-05
Resource Use - fossils [MJ]	2.17E+02	9.27E+03	7.51E+01	3.32E+02	-2.12E+03	7.77E+03	3.80E+03
Biodiversity [species*year]	2.41E-03	7.84E-06	-3.59E-05	1.02E-07	-5.38E-04	1.84E-03	7.38E-07

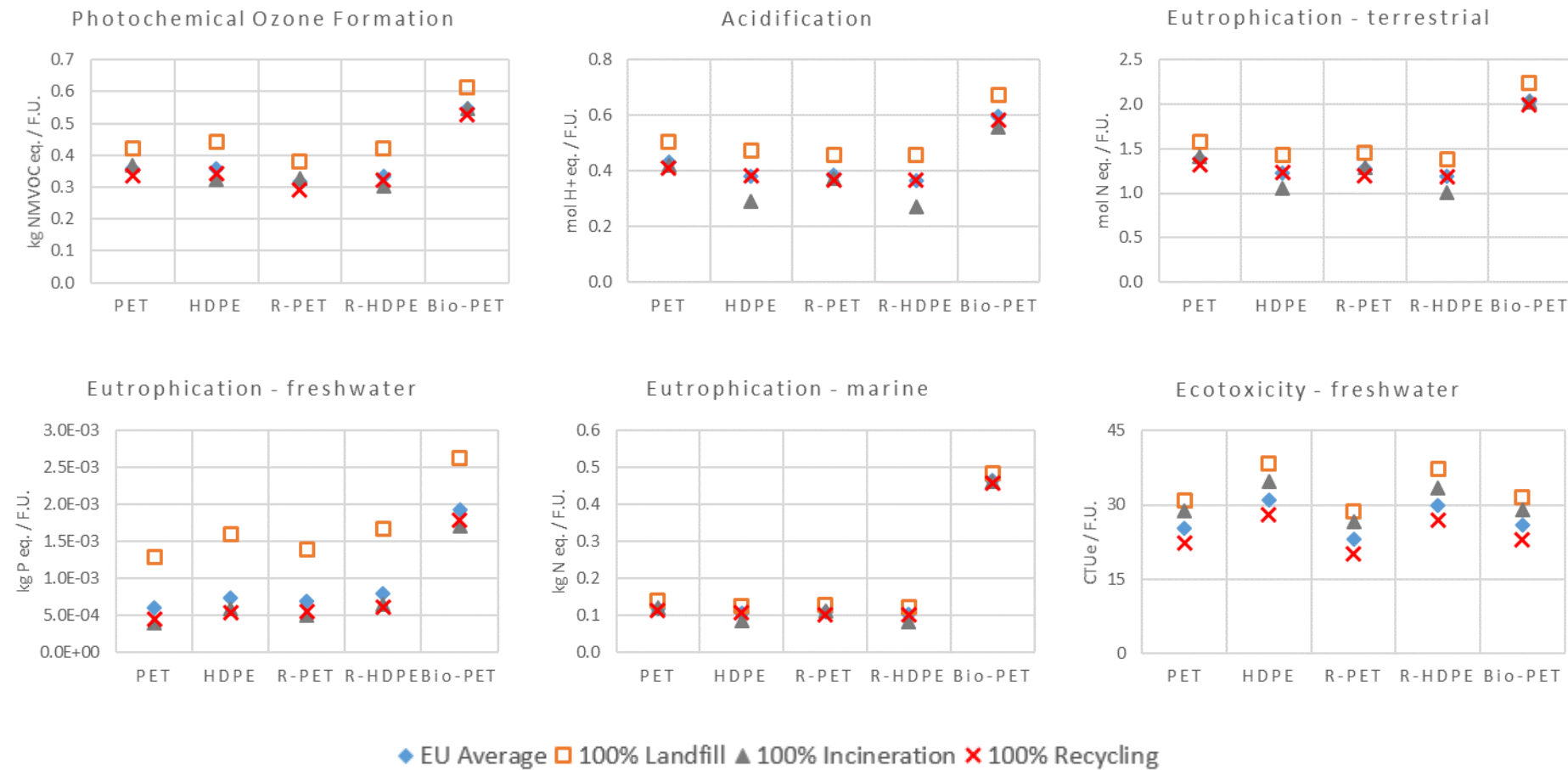
2 (<sup>1</sup>) Accounts for the contribution of biogenic carbon not released after 100 years from landfilling of PEF bottles.

1

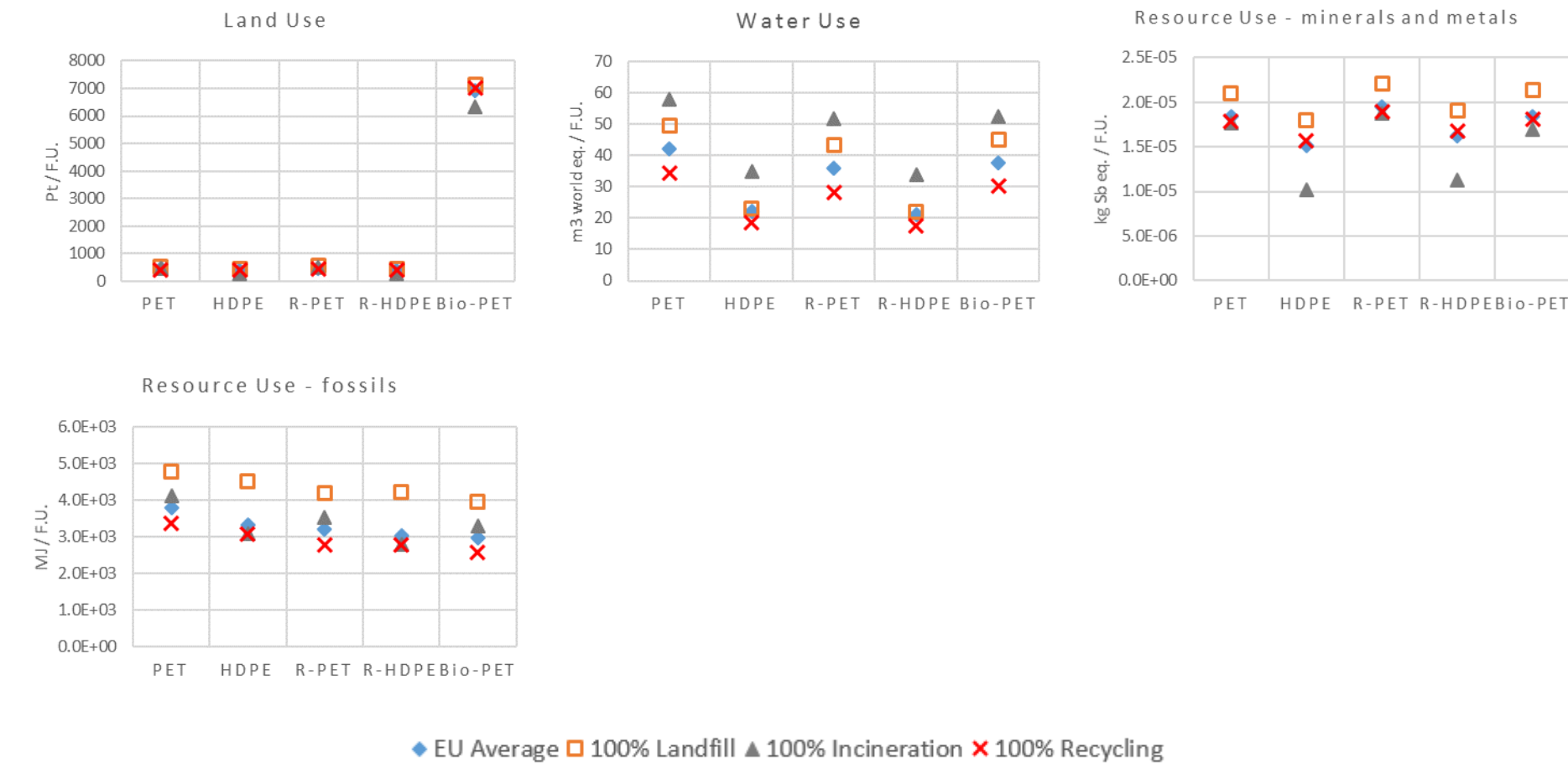


**Figure 4.12.** Potential impact of beverage bottles LCA scenarios for different End of Life options, for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity – non-cancer, Particulate Matter and Ionising Radiation. The Climate Change impact of 100% landfilling scenarios denoted with “C bio EoL” accounts for the contribution of biogenic carbon not released after 100 years from landfill deposition of bio-based beverage bottles.

1



2 **Figure 4.13.** Potential impact of beverage bottles LCA scenarios for different End of Life options, for the categories of Photochemical Ozone Formation,  
3 Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater.  
4



**Figure 4.14.** Potential impact of beverage bottles LCA scenarios for different End of Life options, for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils.

## 4.6 Additional Environmental Information

This section presents additional environmental impacts or aspects going beyond the default impact categories considered in the present method, which reflect official categories adopted in the PEF context. Additional environmental issues addressed include the impact from indirect Land Use Change (iLUC) on Climate Change, potential Biodiversity impacts, as well as the contribution of the investigated beverage bottles scenarios to macro-plastics formation at End of Life (including product litter) and to micro-plastics generation throughout the supply chain.

### 4.6.1 iLUC impact

Table 4.17 presents the estimated contribution of GHG emissions from iLUC induced by bio-based feedstock supply to the potential Climate Change impact of bio-based beverage bottles.

**Table 4.17.** iLUC contribution to the potential Climate Change impact of beverage bottles LCA scenarios.

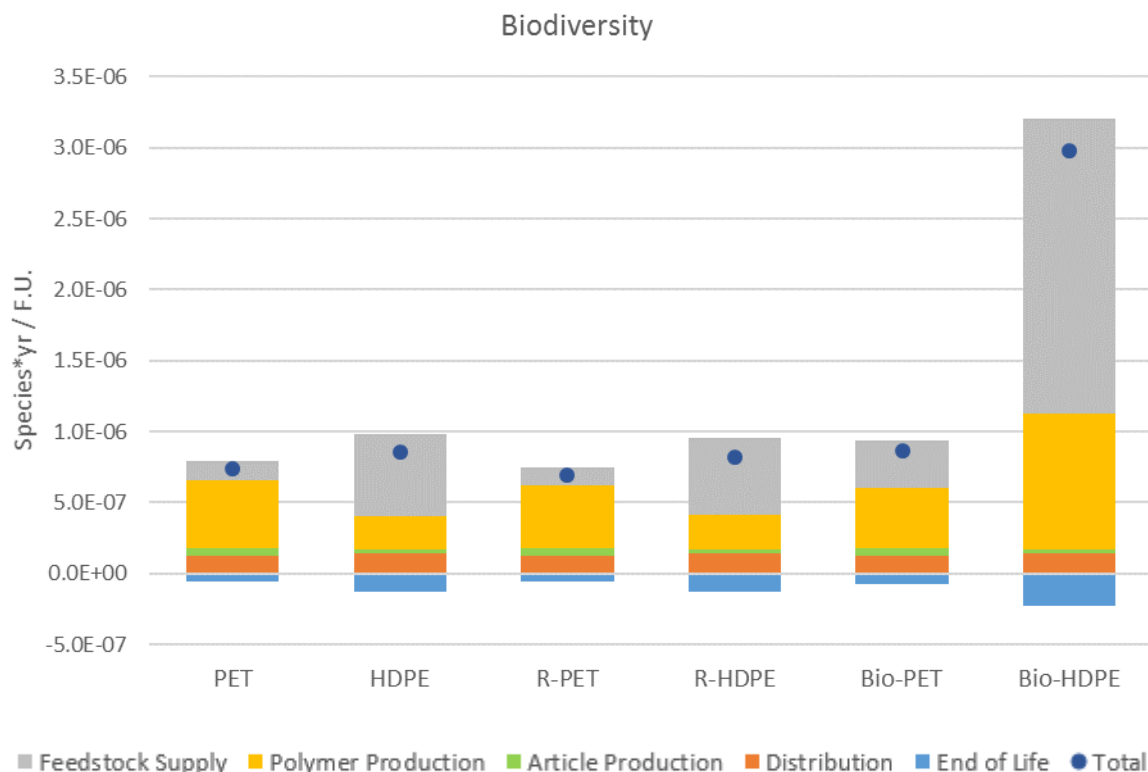
Scenario / Polymer	iLUC Climate Change impact [kg CO <sub>2</sub> eq/FU]	Total Climate Change impact (incl. iLUC) <sup>(1)</sup> [kg CO <sub>2</sub> eq/FU]
S1 – PET	-	(193)
S2 – HDPE	-	(139)
S3 – R-PET	-	(172)
S4 – R-HDPE	-	(133)
S5 – Bio-PET	4.53	207 (202)
S6 – Bio-HDPE	32.0	412 (380)
S7 – PEF	7.71	554 (546)

<sup>(1)</sup> Values in parenthesis refer to the Climate Change impact without the iLUC contribution.

### 4.6.2 Biodiversity impacts

Estimated potential Biodiversity impacts of the investigated scenarios are presented in Figure 4.15. It is reminded that the assessment is performed through an endpoint-level indicator relying on impact assessment methods that for some of the underlying midpoint impact drivers differ from those recommended in the present method.

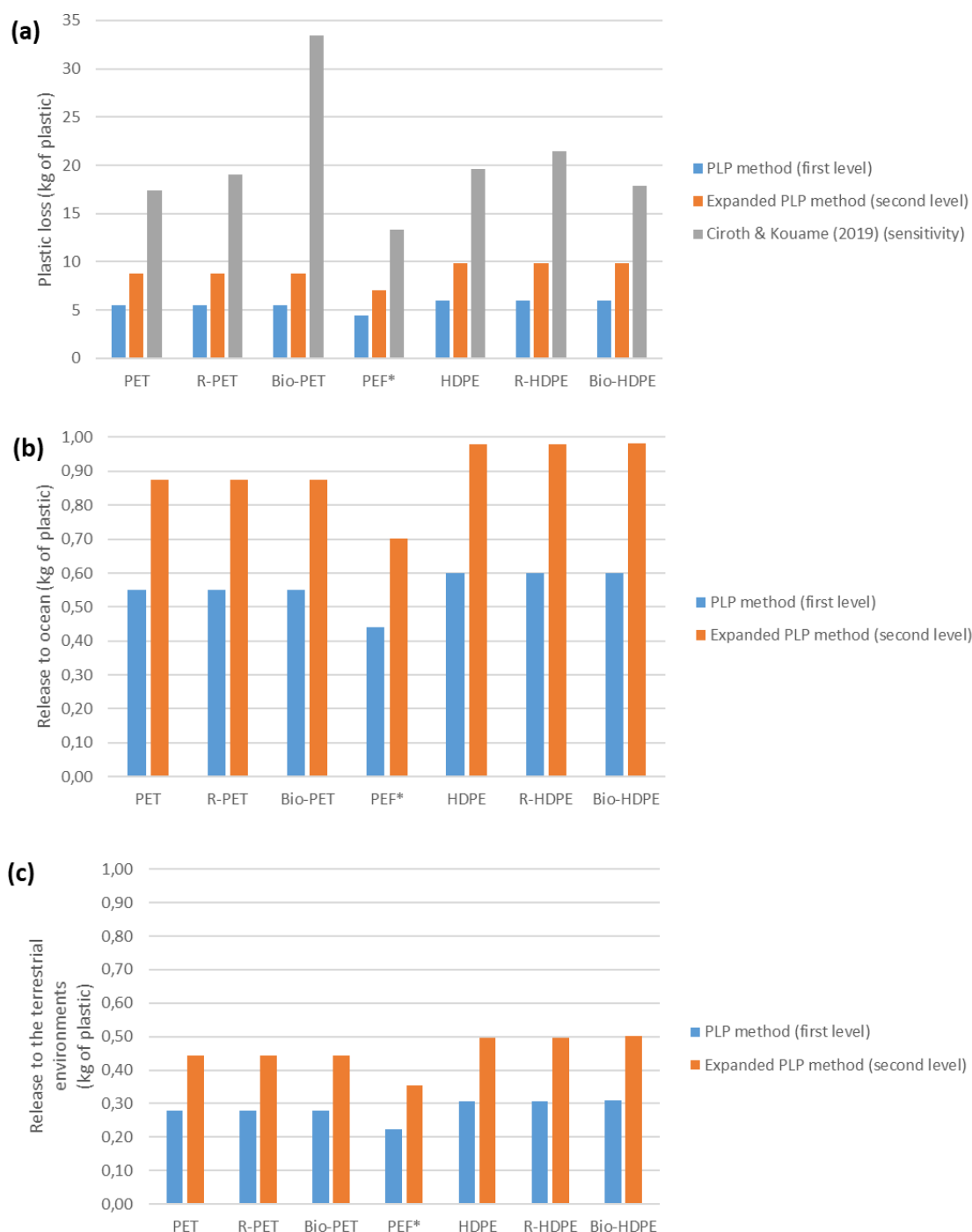




**Figure 4.15.** Potential biodiversity impact of beverage bottles LCA scenarios, expressed as potential loss of species per year.

### 4.6.3 Macro- and micro-plastics generation

Figure 4.16 shows the total plastic loss, release to ocean and release to the terrestrial environment estimated for the assessed beverage bottle scenarios, considering the whole product life cycle. Estimates obtained by applying all the three different explored approaches are presented, including the total loss and release to ocean/terrestrial environment calculated through both the PLP method (first level) and the Expanded PLP method (second level), as well as the alternative estimate of the total supply-chain loss of plastics from applying the sensitivity approach by Ciroth and Kouame (2019).

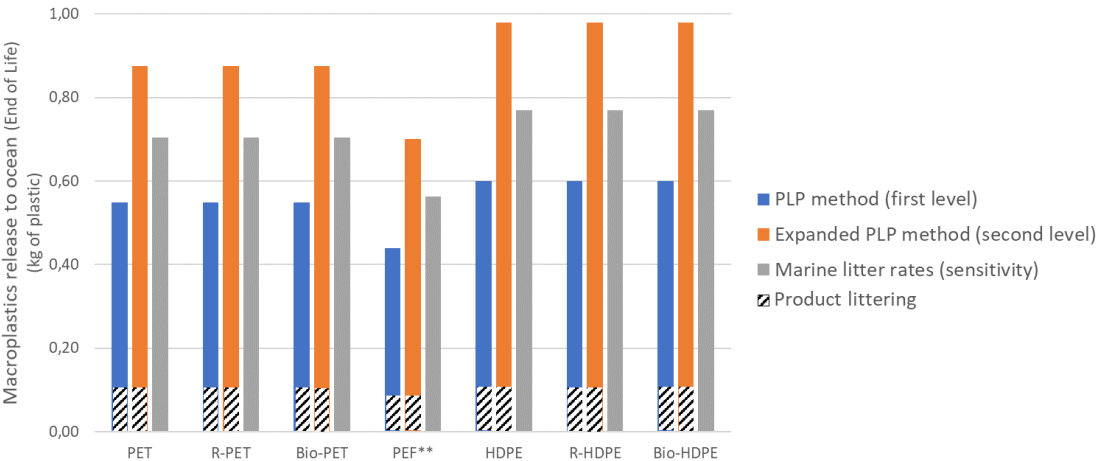


(\*) PEF is an emerging polymer and the related results need to be interpreted with caution, although they are not significantly affected by the lower maturity level of some of the underlying production processes.

**Figure 4.16.** Total plastic loss (a), release to ocean (b) and release to the terrestrial environment (c) estimated for beverage bottles LCA scenarios with different approaches. The contribution of both macro- and micro-plastics is included, considering the whole product life cycle.

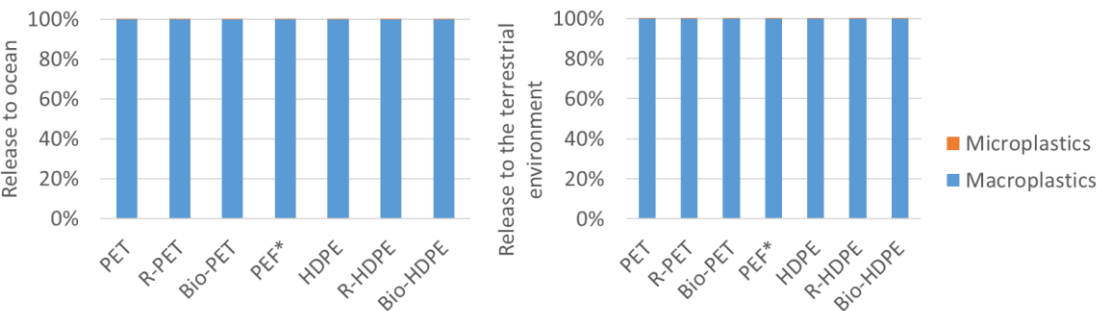
Regarding the sole release of macro-plastics to ocean at End of Life (from product littering and waste mismanagement), Figure 4.17 presents the contribution of each beverage bottles scenario, estimated with the three different explored approaches: the *PLP method* (first level), the *Expanded PLP method* (second level), and the sensitivity approach based on product-specific marine litter rates from observed beach litter in the EU. Note that the latter do not differentiate the type of source (e.g. littering and waste mismanagement), but inherently takes into account all the possible sources that may

ultimately contribute to macro-plastic release to ocean at the European level (still with a higher degree of uncertainty).

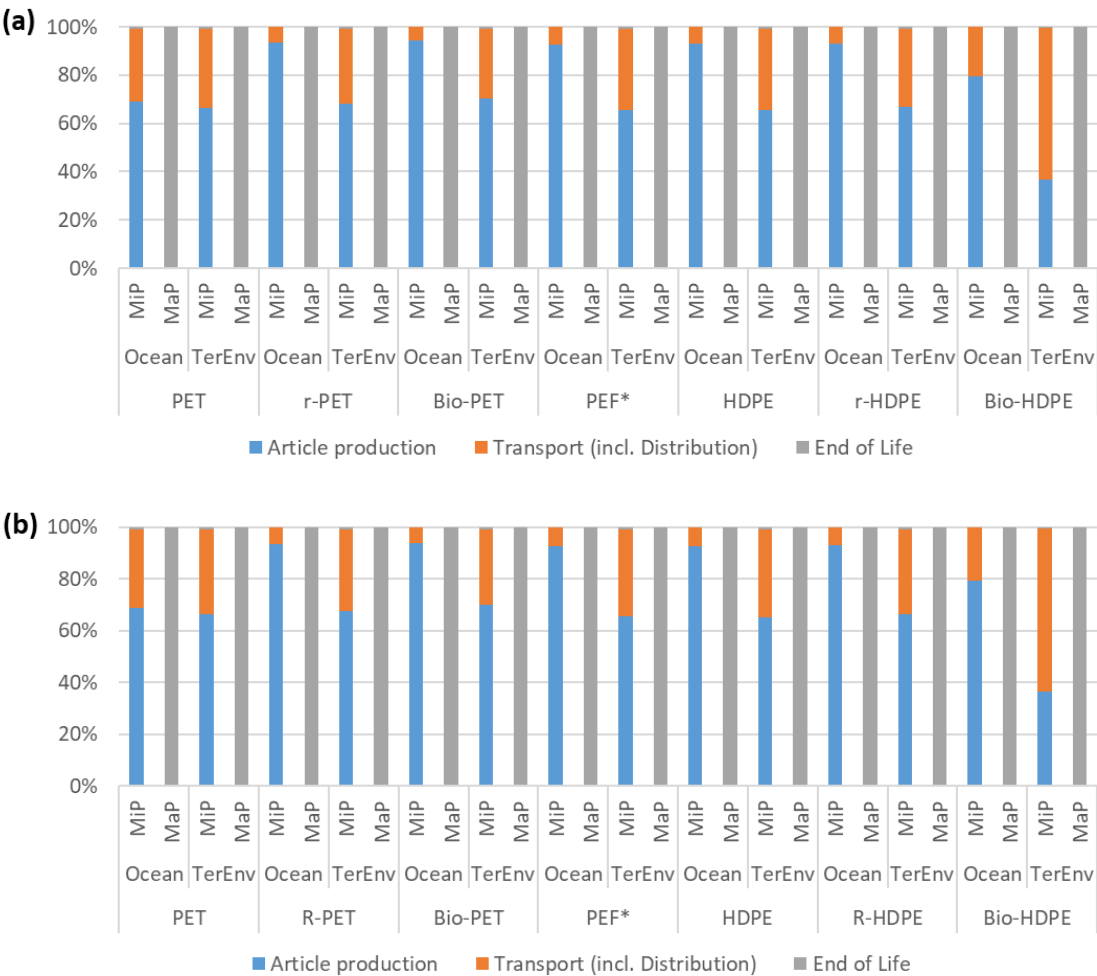


(\*) PEF is an emerging polymer but these results are not affected by the lower maturity level of some of the underlying production processes.

**Figure 4.17.** Macro-plastics release to ocean at End of Life, estimated with different approaches. As for the share between the release of macro- and micro-plastics, Figure 4.18 shows as an example the relative contribution of the two categories to the total release to ocean and to the terrestrial environment estimated through the *PLP method* (first level). Note that the results are similar to those obtained when applying the *Expanded PLP method* (second level), as further discussed in Section 4.7.7.



**Figure 4.18.** Contribution of macro- and micro-plastics to the total plastic release to ocean and to the terrestrial environment estimated with the *PLP method* for each beverage bottles LCA scenario. The contribution of the different lifecycle stages to the total macro- and micro-plastics release to ocean and to the terrestrial environment of the assessed beverage bottles scenarios has been finally evaluated. The results of this “hotspot analysis” are reported in Figure 4.19 for both the *PLP method* and the *Expanded PLP method*. Results are not shown for the other (sensitivity) approaches as they either cover only plastic losses or only a part of the product life cycle (i.e. End of Life).



(\*) PEF is an emerging polymer and the related results need to be interpreted with caution, although they are not significantly affected by the lower maturity level of some of the underlying production processes.

**Figure 4.19.** Contribution of life cycle stages to the total macro-plastics (MaP) and micro-plastics (MiP) release to ocean (Ocean) and to the terrestrial environment (TerEnv) estimated for beverage bottles LCA scenarios with (a) the PLP method (first level) and (b) the Expanded PLP method (second level).

## 4.7 Interpretation

In the interpretation of case study results, most relevant impact categories and lifecycle stages of the studied systems are firstly identified (4.7.1 and 4.7.2, respectively). Scenario impacts are then compared, considering both characterised LCIA results (4.7.3) and total normalised and weighted impacts (4.7.4). Results obtained by individually applying each viable End of Life option are separately discussed in Section 4.7.6, while the effects of iLUC and the contribution to macro- and micro plastics generation are addressed in Sections 4.7.5 and 4.7.7, respectively. Finally, the results of the sensitivity analysis on relevant parameters and assumptions are presented (4.7.8).

Most relevant processes were not identified, since it was not possible to achieve a similar level of vertical disaggregation for foreground processes included in the life cycle models of the analysed scenarios (e.g. a higher disaggregation could be achieved for bio-based HDPE and PEF bottles, but this was not the case of other scenarios). Therefore, the identification of most relevant processes cannot be carried out consistently across all scenarios, and a more detailed investigation for specific scenarios would not be meaningful. The identification of most relevant elementary flows was also not undertaken, as this would require prior identification of most relevant processes. Note,

however, that any company, organisation or any other supply chain actor applying the present method shall proceed with the identification of both most relevant processes and elementary flows.

#### **4.7.1 Identification of the most relevant impact categories**

Table 4.18 shows the most relevant impact categories identified for each beverage bottles scenario, based on normalised and weighted impacts, according to the approach outlined in the Method (Report I, Section 6.2.1. Relevant categories were identified as those that cumulatively contribute to at least 80% of the total normalised and weighted impact of the considered impact categories, excluding toxicity-related ones (these being still based on characterisation factors implemented in the EF 2.0 impact assessment methods, and hence not yet updated based on REACH data<sup>32</sup>). 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 fossil-based PET and HDPE bottles (both virgin and recycled), only two impact categories are identified as most relevant, i.e. Climate Change and Resource Use – fossils. Climate Change is the most relevant category for virgin and recycled PET bottles, while for virgin and recycled HDPE bottles, Resource Use – fossils is the most relevant one. These two categories are also the most relevant for PEF bottles, although in this case also Particulate Matter and Acidification are included in the ranking, yet with a lower contribution (13.7% and 4.3%).

In the case of bio-based PET and HDPE bottles, Particulate Matter is the most relevant category, followed by Climate Change and Resource Use – fossils (Bio-PET), or Eutrophication – marine (Bio-HDPE).

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<sup>32</sup> According to the latest version of the Product Environmental Footprint Category Rules Guidance (EC, 2018), toxicity-related impact indicators based on these characterisation factors shall be excluded from the procedure to identify most relevant impact categories.

**Table 4.18.** Most relevant impact categories identified for beverage bottles LCA scenarios and related contribution to the total normalised and weighted scenario impact.

S1 - Fossil-based PET		S2 – Fossil-based HDPE		S3- Recycled PET	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Climate Change	42.0%	Resource Use - fossils	42.9%	Climate Change	42.9%
Resource Use – fossils	39.6%	Climate Change	37.7%	Resource Use - fossils	38.2%
Acidification	3.9%	Particulate Matter	4.4%	Acidification	4.0
<b>Total</b>	<b>85.5%</b>	<b>Total</b>	<b>85.0%</b>	<b>Total</b>	<b>85.1%</b>
S4 – Recycled HDPE		S5 – Bio-based PET		S6 – Bio-based HDPE	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Resource Use – fossils	41.8%	Particulate Matter	34.3%	Particulate Matter	65.8%
Climate Change	38.4%	Climate Change	29.0%	Climate Change	14.0%
Particulate Matter	4.5%	Resource Use - fossils	20.3%	Eutrophication - marine	4.0%
<b>Total</b>	<b>84.7%</b>	<b>Total</b>	<b>83.6%</b>	<b>Total</b>	<b>83.8%</b>
S7 – PEF					
<i>Impact category</i>	<i>Contrib.</i>				
Climate Change	37.7%				
Resource Use - fossils	25.7%				
Particulate Matter	13.7%				
Acidification	4.3%				
<b>Total</b>	<b>81.4%</b>				

#### 4.7.2 Identification of most relevant life-cycle stages

Table 4.19 shows the most relevant lifecycle stages in the relevant impact categories identified in Section 4.7.1, and the associated contribution, quantified according to the rules detailed in the Method (Report I, Section 6.2.2). Most relevant stages include those that together contribute to at least 80% of the total characterised lifecycle impact in the specific category, and are highlighted in yellow in Table 4.19. For a more exhaustive picture, the contribution of all lifecycle stages is reported.

For fossil-based PET bottles (both virgin and partially recycled), Polymer Production is identified as the most relevant stage, followed by Distribution (in Climate Change) 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 (recycled HDPE). In Resource Use – fossils, Feedstock Supply is the only relevant

stage for these alternatives, where it contributes, alone, to more than 90% of the overall impact.

For bio-based PET bottles, Polymer Production and Feedstock Supply are the most relevant stages in Climate Change and Resource Use – fossils, while in Particulate Matter the stage of Feedstock Supply contributes, alone, to almost 95% of the overall impact. A similar situation is observed also for bio-based HDPE bottles, where Feedstock Supply is alone responsible for 75-89% of the total impact in all of the three most relevant categories, reflecting the relatively high specific consumption of sugarcane for polymer production (28.6 kg of sugarcane per kg of HDPE).

In the case of PEF bottles, Polymer Production dominates the overall impact in all the identified most relevant categories. However, this is affected by the relevant impact associated with HMF production, which is likely overestimated compared to any future industrial production at a larger scale (see Section 4.4.2.3.3).

**Table 4.19.** Contribution of life cycle stages to the total characterised impacts of the most relevant categories identified for beverage bottles LCA scenarios. Most relevant stages (i.e. those contributing to more than 80% of the total impact) are highlighted in yellow.

<b>S1 – Fossil-based PET</b>					
<b>Climate Change</b>		<b>Resource Use - fossils</b>		<b>Acidification</b>	
<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	72.9%	Polymer Production	56.5%	Polymer Production	60.6%
Distribution	16.5%	Feedstock Supply	52.1%	Distribution	25.5%
Feedstock Supply	8.6%	Distribution	10.9%	Feedstock Supply	19.4%
Article Production	7.0%	Article Production	5.5%	Article Production	8.8%
End of Life	-5.1%	End of Life	-25.0%	End of Life	-14.3%
<b>S2 – Fossil-based HDPE</b>					
<b>Resource Use - fossils</b>		<b>Climate Change</b>		<b>Particulate Matter</b>	
<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	90.5%	Polymer Production	46.6%	Polymer Production	61.1%
Polymer Production	27.5%	Feedstock Supply	25.6%	Distribution	30.1%
Distribution	13.7%	Distribution	25.1%	Feedstock Supply	29.9%
Article Production	3.6%	Article Production	5.1%	Article Production	7.8%
End of Life	-35.3%	End of Life	-2.4%	End of Life	-28.8%
<b>S3 – Recycled PET</b>					
<b>Climate Change</b>		<b>Resource Use - fossils</b>		<b>Acidification</b>	
<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	70.8%	Polymer Production	56.9%	Polymer Production	58.8 %
Distribution	18.5%	Feedstock Supply	53.2%	Distribution	28.5%
Feedstock Supply	8.5%	Distribution	12.9%	Feedstock Supply	18.9%
Article Production	7.9%	Article Production	6.5%	Article Production	9.8%
End of Life	-5.7%	End of Life	-29.6%	End of Life	-16.0%

S4 – Recycled HDPE					
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	91.2%	Polymer Production	46.2%	Polymer Production	61.2%
Polymer Production	28.6%	Distribution	26.2%	Distribution	31.7%
Distribution	15.0%	Feedstock Supply	24.8%	Feedstock Supply	29.3%
Article Production	3.9%	Article Production	5.3%	Article Production	8.2%
End of Life	-38.7%	End of Life	-2.6%	End of Life	-30.3%
S5 – Bio-based PET					
Particulate Matter		Climate Change		Resource Use - fossils	
<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	94.4%	Polymer Production	59.4%	Polymer Production	60.8%
Polymer Production	4.8%	Feedstock Supply	25.2%	Feedstock Supply	50.1%
Distribution	1.9%	Distribution	15.8%	Distribution	13.9%
Article Production	0.9%	Article Production	6.7%	Article Production	7.0%
End of Life	-2.0%	End of Life	-7.1%	End of Life	-31.9%
S6 – Bio-based HDPE					
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	88.9%	Feedstock Supply	75.2%	Feedstock Supply	87.5%
Polymer Production	11.1%	Polymer Production	22.5%	Polymer Production	11.5%
Distribution	0.3%	Distribution	9.2%	Distribution	1.5%
Article Production	0.1%	Article Production	1.9%	Article Production	0.2%
End of Life	-0.3%	End of Life	-8.8%	End of Life	-0.7%

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1

<b>S7- PEF</b>					
<b>Climate Change</b>		<b>Resource Use - fossils</b>		<b>Particulate Matter</b>	
<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	110.8%	Polymer Production	119.2%	Feedstock Supply	101.1%
Feedstock Supply	10.2%	Distribution	4.3%	Polymer Production	26.4%
Distribution	4.7%	Feedstock Supply	2.8%	Distribution	1.8%
Article Production	0.5%	Article Production	1.0%	Article Production	-1.0%
End of Life	-26.2%	End of Life	-27.3%	End of Life	-28.3%
<b>Acidification</b>					
<i>Life cycle stage</i>	<i>Contrib.</i>				
Polymer Production	80.2%				
Feedstock Supply	40.2%				
Distribution	5.9%				
Article Production	0.5%				
End of Life	-26.8%				

2

### 3 4.7.3 Interpretation of characterised results

4 In this section, the characterised potential impacts of the assessed scenarios (Figures  
5 4.8-4.10 and Table B.1.8) are compared to evaluate the effects of using an alternative  
6 feedstock or material for a specific fossil-based polymer of reference used in beverage  
7 bottles manufacturing. Therefore, the impacts of partially recycled PET, partially bio-  
8 based PET, and PEF bottles are compared with those of virgin fossil-based PET bottles,  
9 while partially recycled HDPE and bio-based HDPE bottles are compared with fossil-based  
10 HDPE bottles. The impacts of reference scenarios (fossil-based PET and HDPE bottles) are  
11 also initially compared, to provide useful elements to potentially extend the comparative  
12 considerations provided below to any of such reference scenarios.

13 The comparison focuses on the sixteen, default impact categories considered in the  
14 Product Environmental Footprint (PEF) context and in this assessment. Potential impacts  
15 on Biodiversity are discussed separately, as being estimated via an endpoint indicator  
16 that is not recommended in the PEF framework, and partially relying on different impact  
17 assessment methods for the underlying midpoint-level impact categories. In the following  
18 comparison, differences between scenario impacts lower than 10% were not considered  
19 significant, in light of the uncertainty associated with the developed life cycle models and  
20 the applied impact assessment models. Moreover, any comparative considerations  
21 related to Human Toxicity and Ecotoxicity impact categories need to be interpreted in  
22 light of the higher uncertainty of the underlying impact assessment models and results.  
23 The same applies, to a lower extent, also to Water Use and Land Use.

24 Focusing on the relative performance of bottles made of the two fossil-based reference  
25 materials (i.e. PET and HDPE), none of the two is found to be environmentally superior to  
26 the other ones, despite HDPE bottles have a slightly higher (but still comparable) mass  
27 with respect to PET ones (i.e. 27.1 vs 24.8 grams). HDPE bottles are indeed preferable in  
28 9 out of 16 impact categories<sup>33</sup>, mostly due to the lower impact from Polymer Production  
29 and, to a lower extent, Article Production (which compensate for the usually higher

<sup>33</sup> Including Climate Change, Ionising Radiation, Acidification, Eutrophication – terrestrial, Eutrophication – marine, Land Use, Water Use, Resource Use – minerals and metals, and Resource Use – fossils.

Feedstock Supply impact). On the other hand, PET bottles show a lower impact in 5 categories<sup>34</sup>, mainly thanks to the lower impact from Feedstock Supply, and also End of Life in the case of Ozone Depletion. For the remaining categories, the both alternatives show a comparable performance (i.e. relative impact variation lower than 10% or close to it). Note, however, that in the two identified most relevant categories (i.e. Climate Change and Resource Use – fossils for both alternatives), HDPE bottles show an impact reduction equalling 28% and 13%, respectively, compared to PET bottles.

The use of partially recycled PET in bottle manufacturing (i.e. 24%, representing the current average recycled content estimated at the EU level) improves the environmental performance compared to the use of only virgin PET in many impact categories (i.e. all except four). However, for five of them, an impact reduction below 10% is observed (which is not significant), while for the other ones the reduction is limited, being in the range of 10-16%. This modest improvement also reflects the relatively low recycled material content, which could technically reach higher shares (even up to 100% in the case of PET bottles). A sensitivity analysis was thus performed on this assumptions, as detailed in Section 4.7.7.1. It has also to be noted that a worsened performance is achieved in a few (four) impact categories, although only in two of them it was found to be relevant. These include Ozone Depletion (impact increase of nearly 6 times compared to virgin PET bottles), and Eutrophication – freshwater (+15% increase), while for Land Use and Resource Use – minerals and metals the increase is not significant (around 6%). The reason for the huge increase in the Ozone Depletion impact can be mainly found in the emissions of ozone depleting substances from the life cycle of Sodium Hydroxide used during recycling (which is responsible for more than 90% of the increased Polymer Production impacts) and (to a lower extent) of other chemicals used in secondary PET production. In the case of Eutrophication – freshwater, the main responsible is again the increased Polymer Production impact, which is here dominated by waterborne emissions from the treatment of wastewater generated in secondary PET production (38% of total Polymer Production impacts), and from the construction of the respective waste treatment facility (29% of total Polymer Production impacts). However, if this infrastructure process is excluded, the use of R-PET would be comparable with that of fossil-based PET in this impact category.

The effects of incorporating 16% recycled HDPE with virgin HDPE in beverage bottles production are similar to those observed for PET bottles. However, in this case, achieved benefits are lower, due to the lower recycled content. Observed impact reductions range between 2% and 8%, which are not significant, and make the two alternatives (i.e. R-HDPE and virgin HDPE bottles) essentially comparable in the majority of the impact categories. Moreover, similarly to PET bottles, a worsened performance is observed for Ozone Depletion (+103%), due again to the increased Polymer Production impact from the use of Sodium Hydroxide in HDPE recycling. On the other hand, the impact increase observed for Eutrophication – freshwater, Land Use and Resource Use – minerals and metals is, in this case, limited (in the range of 7-9%) and not significant.

Replacing fossil-based MEG with bio-based MEG from Brazilian sugarcane in PET production (i.e. in partially bio-based PET bottles) is not proven to be beneficial for more than half of the assessed impact categories (i.e. 9 over 16). In most of these categories, an impact increase of between 16% and 290% is observed compared to fossil-based PET bottles, with Particulate Matter and Land Use even showing a larger increase (nearly 15 times). For most of the remaining categories (i.e. five), both alternatives are comparable. An impact reduction is only observed for Ozone Depletion (-28%) and Resource Use – fossils (-22%), in this case thanks to the partial replacement of a fossil-based with a bio-based feedstock source. The reason of the overall worsened performance can be found in the increased impact from Feedstock Supply and, in few categories (including Acidification, Eutrophication – terrestrial, and Photochemical Ozone Formation) also of Polymer Production. As reported above, the highest impact increase is observed for Land

<sup>34</sup> Including Ozone Depletion, Human Toxicity - cancer, Human Toxicity - non-cancer, Eutrophication - freshwater, and Ecotoxicity – freshwater.

1 Use and Particulate Matter, due respectively to the incorporation of a “land-based”  
2 monomer (i.e. bio-MEG), and to the release of fine particulate (PM 2.5) from sugarcane  
3 cultivation. While the actual origin of these particulate emissions could not be  
4 investigated within the applied cultivation dataset (aggregated), they are likely  
5 associated with the pre-harvest burning practice, which was applied to 45% of harvested  
6 sugarcane. However, a sensitivity analysis was performed on this assumption, as detailed  
7 in Section 4.7.7.2. A relevant impact increase is also observed for Eutrophication –  
8 marine (+289%), Eutrophication – freshwater (+220%), and Human Toxicity – non-  
9 cancer (+134%) mostly due to the increased impact from Feedstock Supply. Responsible  
10 for this increase are freshwater emissions of nitrate (Eutrophication – freshwater) and  
11 phosphate (Eutrophication – marine), both likely originating from fertiliser application. In  
12 Human Toxicity – non-cancer, Cadmium and Lead emissions to air dominate Feedstock  
13 Supply impacts. For the other categories, the increase is lower, but still relevant,  
14 especially for Photochemical Ozone Formation (+53%), Eutrophication – terrestrial  
15 (+47%), and Acidification (38%). In the case of Eutrophication and Acidification,  
16 Ammonia emissions (again, likely originating from fertiliser application) dominate the  
17 increased Feedstock Supply impact.

18 The use of sugarcane-derived bio-Ethylene in HDPE production substantially increases  
19 the impact of bottles made of this material compared to the use of fossil-based Ethylene.  
20 An exception is Resource Use – fossils, which is decreased thanks to the reduced use of  
21 fossil feedstock sources. The main responsible for this overall worsened performance is  
22 the increased Feedstock Supply impact, which also reflect the high specific sugarcane  
23 consumption for Bio-HDPE production (equalling 28.6 kg per kg of HDPE). The highest  
24 impact increases are observed for Land Use (increased land demand for biomass  
25 growth), Particulate Matter (pre-harvest burning practice in sugarcane cultivation),  
26 Human Toxicity – non-cancer (heavy metal emissions from farming), and both  
27 Eutrophication – marine and Eutrophication – freshwater (nutrient emissions from  
28 cultivation). Relevant increases also take place in Eutrophication – terrestrial (due again  
29 to nutrient emissions), Photochemical ozone formation (CO and NO<sub>x</sub> emissions with  
30 unspecified origin), and Acidification (NO<sub>x</sub> emissions), where also the increased transport  
31 impact of bioethanol from Brazil to Europe plays a role beyond emissions from sugarcane  
32 cultivation. For the remaining categories, the increase is lower, but still significant,  
33 ranging between 84% (Ionising Radiation) to 297% (Resource Use – minerals and  
34 metals). Note that in the case of Ozone Depletion, Ionising Radiation, and Water Use, the  
35 main responsible is the increased impact from Polymer Production, rather than of  
36 Feedstock Supply. In particular, Ozone Depletion impacts are dominated by chemicals  
37 and natural gas used during polymerisation, while production of electricity used in the  
38 same process dominates the Ionising Radiation impact. For Water Use, the largest  
39 contribution is provided by water consumption in Ethanol production. In this respect, it  
40 has to be noted that values of water consumption considered in the adjusted ecoinvent  
41 dataset applied to model Ethanol production are higher than those reported in the  
42 inventory developed for Braskem (ACV Brasil, 2017) applied to downstream conversion  
43 and polymerisation processes. For consistency reasons, no changes were made, however,  
44 to default values in the applied ecoinvent dataset.

45 At the current level of development of the underlying production technology, the  
46 environmental profile of PEF bottles (Tables 4.12 and B.1.11) is not comparable with that  
47 of bottles made of fossil-based PET (i.e. the reference material that PEF intends to  
48 replace). Despite the potential material savings estimated and accounted in the  
49 assessment for PEF bottles (i.e. 20% mass reduction compared to PET bottles<sup>35</sup>), the  
50 potential impacts associated with the use of this polymer as a bottle material are much  
51 higher (from 2 to 120 times) than those associated with the use PET across all impact  
52 categories. To achieve comparable performances, a substantial impact reduction between  
53 51% and 99% should be pursued, as better detailed in Table 4.20. However, these  
54 results must also be read in light of the nature of the inventory data applied to the

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<sup>35</sup> Based on improved barrier and mechanical properties, as detailed in Section 4.2.

modelling of the synthesis of HMF (a precursor of FDCA monomer), which are derived from process simulation and do not reflect full-scale production data from real facilities (although partial process optimisation via energy integration is considered in the generation of such data). Note that potential impacts of PEF bottles were also recalculated by excluding the contribution of refrigeration energy, which had been modelled through a proxy dataset related to cooling energy supply, potentially overestimating the respective burdens and impacts. However, the resulting environmental profile is still unfavourable, with PEF bottles showing an impact between 1 and more than 100 times higher than PET bottles. A similar situation is observed also when the potentially overestimated contribution of both cooling and refrigeration energy is excluded, with an impact increase that for most categories falls in the same range as above. Exceptions are the categories of Resource Use – fossils (where a 40% impact reduction is observed), and Climate Change (where both alternatives are comparable). It must be noted, however, that the considered scenario is optimistic, as it totally disregards a share of the energy demand that would take place in a real production facility.

**Table 4.20.** Required improvement in the environmental performance of PEF beverage bottles to achieve the same performance as PET bottles (reference material).

Impact category	Reduction (%)
Climate Change	64.6
Ozone Depletion	99.2
Human Toxicity - cancer	62.7
Human toxicity - non-cancer	93.3
Particulate matter	91.8
Ionising Radiation	59.4
Photochemical Ozone Formation	62.7
Acidification	71.1
Eutrophication - terrestrial	70.8
Eutrophication - freshwater	98.6
Eutrophication - marine	86.1
Ecotoxicity - freshwater	96.6
Land Use	96.0
Water Use	79.6
Resource Use - mineral and metals	97.9
Resource Use - fossils	51.2

Focusing on estimated potential impacts on biodiversity (Figure 4.15), both partially recycled PET and HDPE bottles are essentially aligned with their virgin, fossil-based counterpart (the considered recycled content is limited and supply chain impact drivers are similar). Conversely, partially bio-based PET and bio-based HDPE bottles involve a higher biodiversity impact, especially in the case of Bio-HDPE ones (which are fully bio-based). This is due to the increased impact from Feedstock Supply, and the resulting land occupation for sugarcane cultivation. In the case of Bio-HDPE bottles, also an increased polymer production impact is observed (and dominated by the impact of biogenic CO<sub>2</sub> emissions from sugarcane conversion to Ethanol). It has to be reminded, however, that direct potential biodiversity (ecosystem) impacts from oil leakage are not quantified for bottles made of fossil-based polymers considered as a reference (although emissions from leakage per unit of oil supplied are reported to be quite small; see Section 4.4.1.1).

As a last comment, it is noted how the Climate Change impact of bio-based, non-biodegradable beverage bottles (i.e. Bio-PET, Bio-HDPE and PEF bottles) is only slightly reduced if the contribution of non-released biogenic carbon after 100 years from landfilling is taken into account. Indeed, a decrease between 2% (Bio-PET) and 7.5% (Bio-HDPE) is observed, while for PEF the reduction equals 3%. This is partly explained by the modest share of landfilling in the applied EU-average scenario (19%) and, in the case of bio-based PET, by the only partial biogenic carbon content in the polymer (i.e. 12.5%).

#### **4.7.4 Interpretation of normalised and weighted results**

This section briefly compares the investigated scenarios based on the respective total impact scores, calculated by aggregating normalised and weighted impact assessment results across all impact categories (Table B.1.10 in Annex B.1). The comparison is carried out according to the same criteria outlined in Section 4.7.3 in terms of contrasted beverage bottles scenarios, and considering significant only impact differences larger than 10%. While affected by greater uncertainty (from the calculation and application of normalisation and weighting factors), total impact scores provides a more immediate and synthetic indication of the overall environmental performance of the analysed product scenarios. This facilitates the interpretation compared to looking at characterised results from several impact categories. However, such estimates are inevitably affected by value choices underlying the definition of weighting factors aimed at establishing an order of relevance for the different impact categories in a European decision context.

The comparison among total normalised and weighted scenario impacts reflects, with some deviations, the overall picture of results emerged when focusing on characterised impact assessment results in single impact categories, and can be summarised in the following points:

- Fossil-based HDPE bottles show a moderately better performance compared to fossil-based PET bottles (20% reduction in the total impact), consistently with the results obtained for the two most relevant impact categories (Climate Change and Resource Use – fossils), where an impact reduction equal to 28% and 13% was respectively observed.
- Partially recycled PET bottles implies only a limited reduction in the total impact compared to their virgin counterpart (-13%), reflecting also the moderate recycled content considered as a base case in the assessment (24%). Similarly, the overall environmental performance of partially recycled and virgin HDPE bottles is comparable, in light of the even more reduced recycled material input (16%).
- Bio-based HDPE bottles from Brazilian sugarcane shows a substantially worse performance compared to fossil-based HDPE bottles, with a total impact score that is more than seven times higher than the latter. For bio-based PET bottles, which only partially rely on a sugarcane-derived precursor (i.e. bio-based MEG; 30% of the polymer), a more limited impact increase is observed compared to their fossil-based counterpart, albeit still significant (+53%).

#### **4.7.5 Effects of indirect land use change (iLUC)**

When the contribution of GHG emissions from iLUC is taken into account, only small or moderate changes are identified in the Climate Change impact of bio-based alternatives (i.e. Bio-PET, Bio-HDPE, and PEF bottles) (Figure 4.8 and Table 4.13). A maximum increase by 8.4% is observed for bio-based HDPE bottles, while for bio-based PET bottles the increase is equal to only 2.2%, reflecting the only 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 higher Climate Change impact associated with the use of this material even without considering iLUC.

Overall, accounting for iLUC effects does not affect the relative performance of the assessed scenarios (i.e. their comparison is not affected). However, this outcome needs to be read also in light of the GHG emission factors applied for iLUC in this assessment, which appear to fall in the lower end of the range of values available in the recent literature (see Section 4.4.16.3 of the Method (Report I)). The use of an alternative iLUC model and of the resulting emission factors was explored as a sensitivity analysis (see Section 4.7.8.3).

#### **4.7.6 Alternative End of Life options**

This section discusses characterised scenario results calculated by individually applying each viable End of Life option as an exclusive (100%) End of Life scenario (as presented in Figures 4.12 to 4.14 and Tables B.1.12. and B.1.13 in Annex B.1). The main purpose is to evaluate how the performances of the single beverage bottles scenarios are affected by the change in the applied End of Life option, rather than to mutually compare such scenarios for different End of Life options. However, the results should not be interpreted as a strict comparison among different End of Life alternatives for the studied article, since the evaluation applies a product perspective, and burdens/benefits of selected End of Life options (e.g. recycling) are partitioned between different product life cycles (which prevents from capturing the full implications of having a given waste stream routed to a specific End of Life option). In a waste management system perspective, where allocation is not needed, the total (system-wise) savings associated with the End of Life pathway “100% recycling” would likely be higher than what presented in this report, where a product perspective is applied.

None of the three considered product End of Life scenarios (i.e. mechanical recycling, incineration and landfilling) can be identified as preferable across all the assessed impact categories and beverage bottles scenarios. If Bio-HDPE bottles are excluded, 100% mechanical recycling is the preferable scenario in six impact categories, including Climate Change, Human Toxicity – cancer, Human Toxicity – non-cancer, Ecotoxicity – freshwater, Water Use and Resource Use – fossils. For Bio-HDPE bottles, this is only the case of Human Toxicity – cancer, Ecotoxicity – freshwater, Water Use and Resource Use – fossils. In many other categories, mechanical recycling is comparable with incineration. In Ozone Depletion, the recycling scenario shows the worst performance for all beverage bottles alternatives except for PEF bottles, where the incineration scenario is worst. In Ionising Radiation, recycling is almost comparable with landfilling, which is the least preferable End of Life scenario. For Bio-HDPE bottles, this is also the case of Land Use.

For all types of bottles except PEF-based ones, 100% incineration is the most favourable scenario only in a limited number of categories, including Ozone Depletion, Ionising Radiation and Resource Use – minerals and metals. In the case of HDPE, R-HDPE and Bio-HDPE bottles, incineration is preferable also in the categories of Particulate Matter, Photochemical Ozone Formation, Acidification, Eutrophication – terrestrial, Eutrophication – marine, and Land Use. On the other hand, the incineration scenario shows the highest impact in terms of Climate Change (all alternatives except for Bio-HDPE and PEF bottles) and Water Use.

In many impact categories, 100% landfilling is the worst End of Life scenario for all or most beverage bottles alternatives, in line with the priority order outlined in the “Waste Hierarchy”, which sets disposal as the least preferable option (EC, 2008). Exceptions are Climate Change, Ozone Depletion, and Water Use, where for most beverage bottles alternatives other End of Life scenarios show the highest impact, as discussed above. Overall, in none of the assessed impact categories and beverage bottles scenarios landfilling is preferable. However, if the contribution of non-released biogenic carbon after 100 years from landfilling is taken into account, the 100% landfilling scenario is preferable for bio-based HDPE bottles, and high-ranked after recycling for bio-based PET bottles. In the case of PEF bottles, 100% recycling is still the most favourable scenario, due to the more important savings from avoided virgin polymer production.

#### **4.7.7 Interpretation of results related to macro- and micro-plastics generation**

This section discusses the results presented in Section 4.6.3 on the estimated potential generation of macro- and micro-plastics of the assessed beverage bottles scenarios. The latter are compared according to the same criteria applied to the interpretation of the potential environmental impacts (Section 4.7.3), initially focusing on the results from the two-level approach applied as a base case. The results from the approaches applied as a sensitivity analysis are discussed separately.

Regarding the total loss of plastics (macro- and micro-plastics) and the resulting release to ocean and to the terrestrial environment (Figure 4.16), the two base-case approaches (PLP method and Expanded PLP method) provide similar results when comparing the different beverage bottles scenarios. In both cases, scenarios considering the use of alternative feedstock sources for PET and HDPE bottles show insignificant differences (<1%) in the total loss and release compared to the respective fossil-based counterparts. This is because changing the feedstock does not affect the dominating contribution from macro-plastics loss and release at End of Life (see below for further discussion on this), while variations in micro-plastics generation due to different transport requirements along the life cycle (reflecting different feedstock or polymer origins) only marginally affect the overall results. When the PLP method is applied, HDPE-based scenarios show, on average, a 9% higher plastic loss and release to the environment compared to PET-based ones, which is not considered significant in this context (similarly to differences in environmental impacts). When applying the Expanded PLP method, where the contribution to macro-plastics loss and release from plastic waste exported for recycling to non-EU countries is considered, the average difference between the two sets of scenarios is moderately (albeit not significantly) increased, with HDPE-based scenarios showing a 12% higher loss and release than PET-based ones. This is because a slightly higher recycling rate is applied to HDPE bottles (64%) compared to PET ones (60%), leading to a larger share of product that is exported for recycling to non-EU countries where mismanagement waste shares are higher. On the other hand, bottles consisting of the emerging polymer PEF involve a lower total loss and release compared to fossil-based PET bottles considered as a reference (-20%), and the lowest among the compared alternatives. This is mostly a consequence of the 20% lower amount of polymer required per functional unit, which leads to a proportionally reduced loss and release of plastics from both upstream life cycle stages (micro-plastics) and especially from product End of Life (macro-plastics). Note, however, that the applied methods rely on mass-based indicators (which are affected by differences in bottle mass) rather than quantifying the number of items (or plastic fragments/particles) lost or released to the environment. If this was the case, all scenarios would result in the same macro-plastics generation at End of Life when applying the PLP method, since bottles with a lower mass would not imply a reduction in the contribution of product littering and waste mismanagement (which are based on release rates that are not affected by the type of feedstock or material). In turn, an identical macro-plastics generation at End of Life would imply having a comparable total loss and release, these being mostly determined by macro-plastics (see below). Conversely, HDPE-based scenarios would still show a moderately higher release of bottles to the environment in the Expanded PLP method (second level), due to the slightly different recycling rate (as discussed above).

Comparing the two base-case approaches, the Expanded PLP method results in a higher plastic loss and release to ocean and the terrestrial environment compared to the original PLP method, equalling on average 59% for PET-based scenarios and 62% for HDPE-based ones (Figure 4.16). This is mainly due to an increased contribution of macro-plastics loss at End of Life, which also accounts for mismanagement of plastic waste exported for (sub-standard) recycling outside the EU (where the overall waste mismanagement share is increased by 5-6% compared to the assumption that all plastic waste is recycled in Europe). Conversely, the two additional micro-plastics sources considered in this alternative approach (road markings and marine coatings) does not

significantly affect the results, due to the overall modest contribution of micro-plastics to the total release (in terms of mass).

The total supply-chain loss of plastics estimated based on the approach by Ciroth and Kouame (2019; sensitivity analysis) is between 3 and 3.6 times higher compared to applying the PLP method, with an even higher increase for Bio-PET bottles (6 times). This is likely a consequence of the broader range of processes covered (albeit in a more generic manner) compared to the PLP method (e.g. the contribution of background transport processes is also accounted for). Comparative results among the different beverage bottles scenarios are similar to those obtained in the base-case assessment, with PEF bottles showing the lowest total loss<sup>36</sup>, and with no relevant differences when changing the type of feedstock used for PET or HDPE bottles production. An exception is represented by Bio-PET bottles, which show a larger plastic loss compared to their completely fossil-based counterpart (+91%). However, this result is affected by the approximations performed in the modelling of Bio-PET production for plastic loss calculation, and may not be sufficiently reliable. More in general, the results obtained by applying this alternative approach need to be interpreted with caution, since the life cycle models of some beverage bottles scenarios present different level of (vertical) disaggregation, and the contribution from intermediate or background processes may have not been taken into account consistently. Moreover, approximations in the modelling of the production of bio-based polymers (beyond the case of Bio-PET discussed above) also affect the respective contribution.

Focusing on the relation between total loss and release (which depends on the parameters of the PLP method), the overall release represents a modest share of the loss (i.e. 15% on average). In relative terms, release to ocean accounts for a larger share of the plastic loss (around 10% on average, corresponding to two third of the total release), while release to the terrestrial environment is lower (5% of the loss and one third of the total release). As shown in Figure 4.18 for the PLP method, the total plastic release to both ocean and the terrestrial environment is dominated by the release of macro-plastics at the End of Life (more than 99%), while micro-plastics released throughout the upstream life cycle have only a minor role. Similar results are also observed for the Expanded PLP method (not shown), despite two additional sources of micro-plastics are considered in the latter (i.e. road markings and marine coatings). In both cases, this is a consequence of the prevailing mass of macro-plastics, which is directly related to the mass of bottles required per functional unit (reference flow), by means of higher loss and release rates compared to micro-plastics. Therefore, the mass of macro-plastics is at least one order of magnitude higher than the mass of micro-plastics, which instead depends on the quantity of relevant lifecycle process by means of (much) lower loss and release coefficients (depending on the source).

Due to the prevailing role of macro-plastics, the comparison among the sole macro-plastics release to ocean (from product littering and waste mismanagement) estimated for beverage bottles scenarios with both base-case approaches reflects the results obtained when looking at the total plastic release into such compartment (Figure 4.17). Comparative results are also in line with those of the sensitivity analysis based on marine litter rates, where PEF bottles still show the lowest release, and insignificant differences are observed with the change in feedstock for PET or HDPE bottles. Similarly to the PLP method, scenarios based on the use of these two polymers also show a comparable release, while in the Expanded PLP Method the contribution of HDPE bottles is moderately higher (12% on average, in line with to the increase observed for the total release). However, it must be reminded that all the approaches rely on mass-based indicators, and the resulting trends are thus determined by the mass of material required to fulfil the functional unit. This implies that scenarios relying on lighter bottles provide a lower contribution to macro-plastics generation (and vice versa), which would not be the case if an indicator based on the number of released items was considered. For all scenarios,

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<sup>36</sup> The same comments as above on the implications of using a mass-based indicator need to be taken into account in the interpretation of this result.



emissions from product littering represent only a modest share of the total macro-plastics release, equalling 18% on average for the PLP method and 11% for the Expanded PLP method, and highlighting the important role of waste mismanagement. Comparing the different approaches, the total release estimated with the Expanded PLP method is higher than the estimates from the official PLP method (and the highest in general) due to the additional (but more uncertain) contribution from mismanagement of plastic waste exported for recycling outside the EU. This contribution is also not captured in the estimates based on EU beach litter observations applied as a sensitivity analysis, which are in between those obtained with the two approaches applied as a base case (for reasons that could not be explored further). Note, however, that such estimates only provides information at the level of product category (i.e. bottles) without accounting for specific product properties (e.g. type of polymer) and for the specific supply-chain configuration (e.g. in terms of type and location of waste treatment operations).

With regard to the origin of released plastics (in terms of life cycle stage and source), macro-plastics entirely originate from the End of Life stage (Figure 4.19), as they are only derived from product littering or mismanagement of the product as waste. On micro-plastics release, the relative contribution of the different sources (e.g. loss of plastic pellets and tire abrasion) depends on the characteristics of the supply-chain (e.g. feedstock origin, which affect the contribution from transport across the life cycle), as well as on the mass of polymer required per functional unit (which affect, to different extents, all the considered sources). However, within the limited role of micro-plastics in the overall release to the environment, the most relevant contribution is generally provided by pellet losses from the Polymer and Article Production stages, while transport-related sources have a more restricted role. An exception is Bio-HDPE bottles, where transport emissions are more important due to increased transport throughout the supply chain (although a more disaggregated modelling of the life cycle also affects the relevance of this contribution).

In conclusion, due to the dominant role of macro-plastics loss and release at the End of Life of beverages bottles, a best overall performance is associated to the scenario requiring a lower amount of plastic to fulfil the functional unit (i.e. relying on lighter bottles; mass-driven results). In this context, the use of PEF bottles is the scenario with the lowest plastic loss and release to both ocean and the terrestrial environment, while regardless of the feedstock HDPE bottles involve a slightly higher plastic loss and release than PET bottles (Expanded PLP method), or comparable to the latter (PLP method). The same considerations apply to the sole release of macro-plastics to ocean at End of Life, although in this case the use of an indicator based on the number of items (i.e. bottles) released would provide the same result regardless of the scenario when applying the PLP method or the sensitivity estimates based on marine litter rates. Bottles with different masses would indeed equally contribute to littering and waste mismanagement at End of Life. Conversely, an items-based indicator would still show different results for HDPE bottles in the Expanded PLP method, with a higher release of macro-plastics due to mismanagement of waste exported for recycling in non-EU countries. From a methodological perspective, the different approaches provide similar results when comparing alternative scenarios, apart from few (relevant) discrepancies when the total loss of plastics is estimated with the approach by Ciroth and Kouame (2019), although the results obtained by applying the latter to the present case study are less reliable than the other approaches.

#### **4.7.8 Sensitivity analysis**

A sensitivity analysis has been performed on a number of relevant parameters or assumptions, to evaluate the effects of their variation on the potential impacts of the affected scenario(s), and on the comparative LCA results. The following aspects have been specifically considered:

1. Recycled content in R-PET and R-HDPE bottles;

2. “Slash and burn” rate for sugarcane used as a feedstock for Bio-PET and Bio-HDPE bottles;
3. Feedstock source for bio-based PET bottles (origin and type);
4. iLUC model and factors;

The following sections present the outcome of the sensitivity analysis for each of the aspects above. The results are reported by comparing recalculated impacts of the affected scenario(s) with those of the base case assessment, of the corresponding reference scenario(s), and of the alternative feedstock scenarios evaluated for the same reference material.

#### **4.7.8.1 Recycled content in R-PET and R-HDPE bottles**

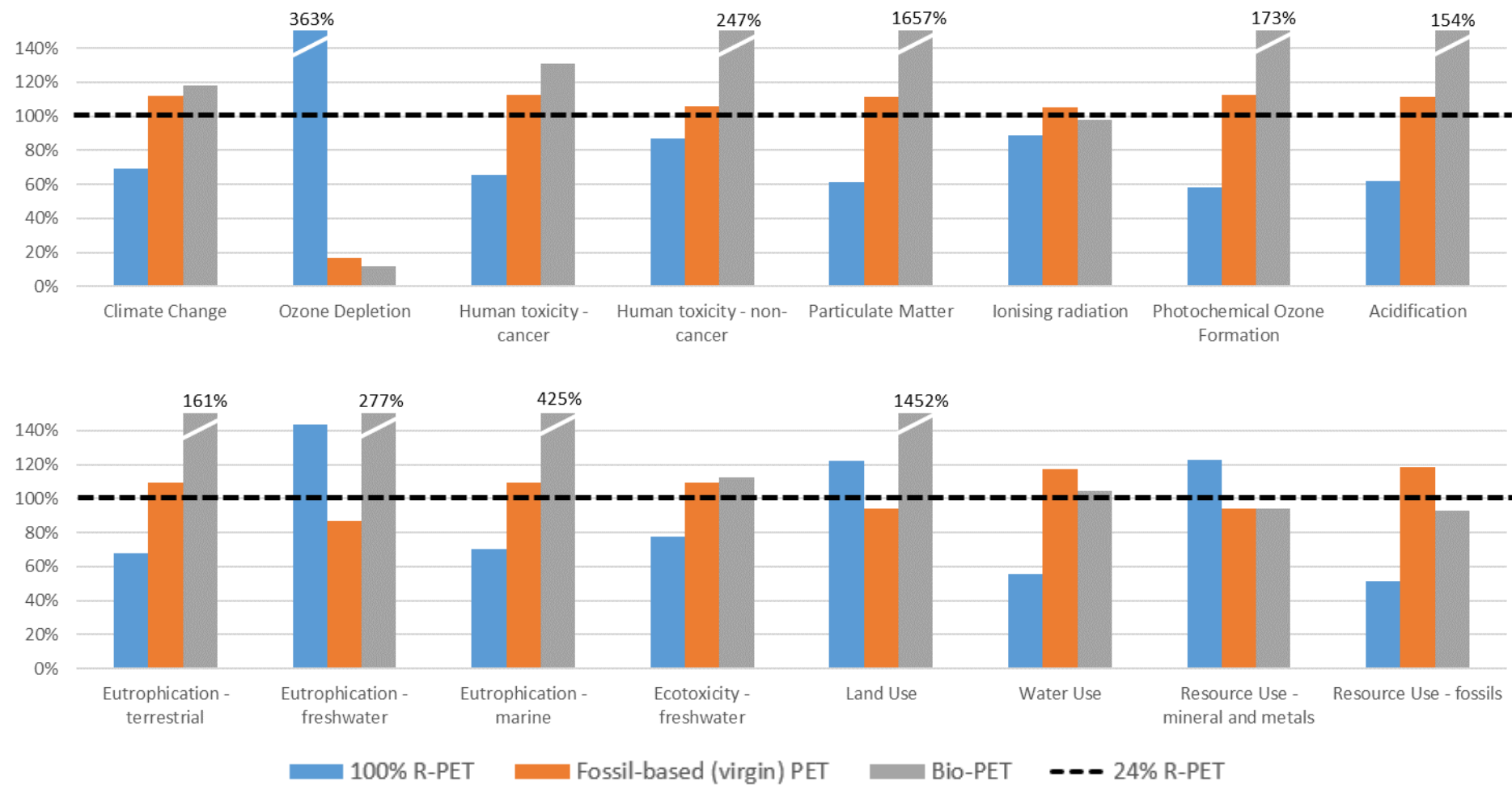
This section explores the effects of increasing the recycled content of PET and HDPE beverage bottles to 100%. As a base case, this parameter was estimated to reflect the current average recycled content at the EU level, equalling 24% for PET bottles and 16% for HDPE bottles. However, from a technical point of view, a higher recycled content could be achieved, potentially reaching 100% for PET bottles and non-food grade HDPE bottles. A sensitivity analysis was thus performed to assess the effects of this possibility.

The results of the analysis are presented in Figure 4.20 for recycled PET bottles and in Figure 4.21 for recycled HDPE bottles. In both figures, scenario impacts are expressed as a function of the impacts of the base case of the affected beverage bottles scenario (i.e. 24% R-PET and 16% R-HDPE bottles, respectively), which is set as a 100% baseline. This allows to show multiple impact categories expressed with different units on the same chart. For graphical reasons and to keep the comparison within reasonable limits, only scenarios having the same reference scenario are shown (i.e. PET-based or HDPE-based scenarios, respectively).

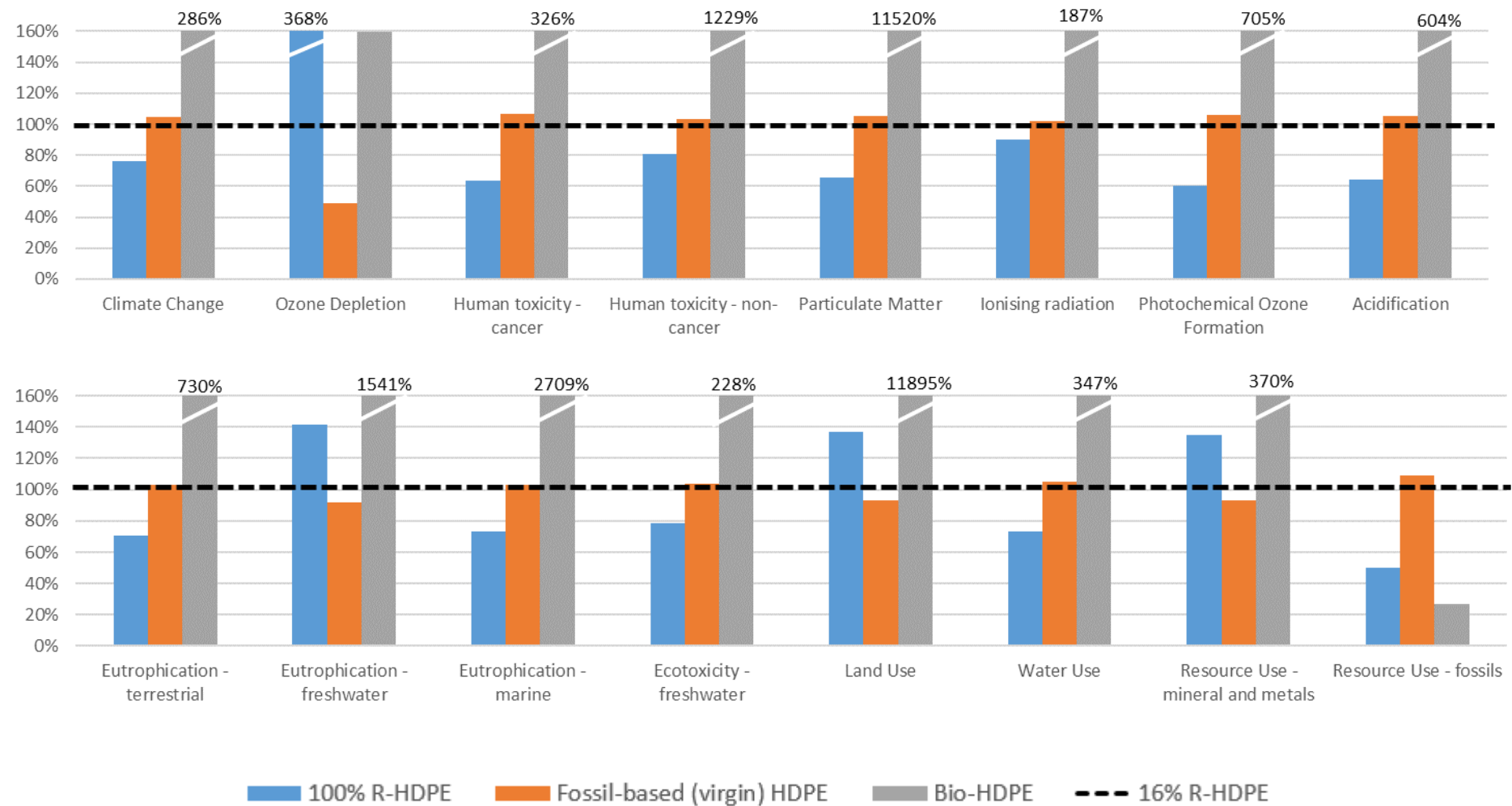
An increased content of secondary material significantly reduces the impact of recycled PET bottles in the vast majority of the assessed categories (i.e. all except four), with a decrease ranging from 11% (Ionising Radiation) to 49% (Resource Use – fossils) compared to the base case. In all these categories, recycled PET bottles now outperform virgin PET and bio-based PET bottles in almost all impact categories, with an impact reduction ranging from 16% to 57% compared to virgin PET bottles. However, with a 100% recycled content, an impact increase is observed in those categories where the use of recycled PET already implied a worsened performance compared to virgin PET<sup>37</sup>, i.e. Ozone Depletion (+263% compared to 24% R-PET bottles), Eutrophication – freshwater (+44%), Land Use (+22%), and Resource Use – minerals and metals (+23%). In these categories, increasing the content of secondary material thus makes the use of recycled PET in bottles manufacturing even less beneficial compared to using its virgin, fossil-based counterpart. Compared to partially bio-based PET bottles, a worsened performance is only observed for Ozone Depletion and Resource Use – minerals and metals, while for the remaining categories, 100% recycled PET bottles are always preferable.

A similar situation is observed also for HDPE bottles, with 100% recycled HDPE bottles outperforming virgin, fossil-based HDPE bottles within the same range of impact categories as above, and with impact reductions between 12% and 54%. Bio-based HDPE bottles are also outperformed in the majority of the impact categories (i.e. all except for Ozone Depletion and Resource Use – fossils).

<sup>37</sup> See Section 4.7.3 for a discussion on the possible reasons for this worsened performance.



**Figure 4.20.** Results of the sensitivity analysis on the recycled material content in PET beverage bottles, increased from 24% to 100%. Scenario impacts are expressed as a percentage of the impacts of the base case of the affected scenario (i.e. 24% R-PET bottles), which is set as a baseline. Note that in some impact categories a part of the results is out of scale and is curtailed.



**Figure 4.21.** Results of the sensitivity analysis on the recycled material content in HDPE beverage bottles, increased from 16% to 100%. Scenario impacts are expressed as a percentage of the impacts of the base case of the affected scenario (i.e. 16% R-HDPE bottles), which is set as a baseline. Note that in some impact categories a part of the results is out of scale and is curtailed.

#### 4.7.8.2 “Slash and burn” rate for sugarcane and feedstock sourcing for Bio-PET

This sensitivity analysis explores the effects of completely removing the “slash and burn” practice in the cultivation of Brazilian sugarcane used as a feedstock for bio-based PET and bio-based HDPE production. 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 “slush and burn” rate was decreased from 45% (assumed as a base case) to 0%.

In addition, this analysis evaluates the effects of replacing Brazilian sugarcane with biomass feedstock sourced in Europe for Bio-PET production. The current EU-average mix of bioethanol crops was considered, which was estimated to include maize (45%), wheat (34%) and sugar beet (21%), based on bioethanol volumes currently obtained from such feedstock sources (ePURE, 2019). These shares applies identically also at the polymer level, since the conversion efficiency of bioethanol to Bio-PET is the same regardless of the feedstock used for Ethanol production. The feedstock substitution was modelled by means of vertically aggregated inventories available in the GaBi database, referring to Bio-PET production from the three types of alternative feedstock considered in this sensitivity analysis. However, since the original dataset for maize-based Bio-PET refers to maize grown in the US, it was adjusted by subtracting the burdens associated with maize cultivation in US, to subsequently add those from maize cultivation in Europe. For this purpose, a specific maize consumption of 0.806 kg/kg Bio-PET was considered<sup>38</sup>, since the actual amount applied in the dataset is not disclosed. To represent the burdens of both US and European maize cultivation, two datasets from the GaBi database were applied, in the attempt to keep consistency with the originally applied dataset (which comes from the same database, despite being unknown). Transoceanic transport of US maize by ship was also initially subtracted, based on an estimated distance of 6000 km (the original assumed distance is also unknown). However, since the results were not significantly different compared to keeping transport included, this change was not ultimately implemented to avoid introducing an additional source of uncertainty.

Figure 4.22 shows the results of both changes for Bio-PET bottles, while Figure 4.23 shows the results obtained for Bio-HDPE bottles. In both figures, scenario impacts are expressed as a function of those of the base case of the affected beverage bottles scenario (i.e. bio-based-PET or bio-based HDPE bottles with Brazilian sugarcane as a feedstock and a “slash and burn” rate of 45%), which is set as a 100% baseline. For graphical reasons, and to keep the comparison within reasonable limits, only scenarios having the same reference scenario are shown (i.e. PET-based or HDPE-based bottles only).

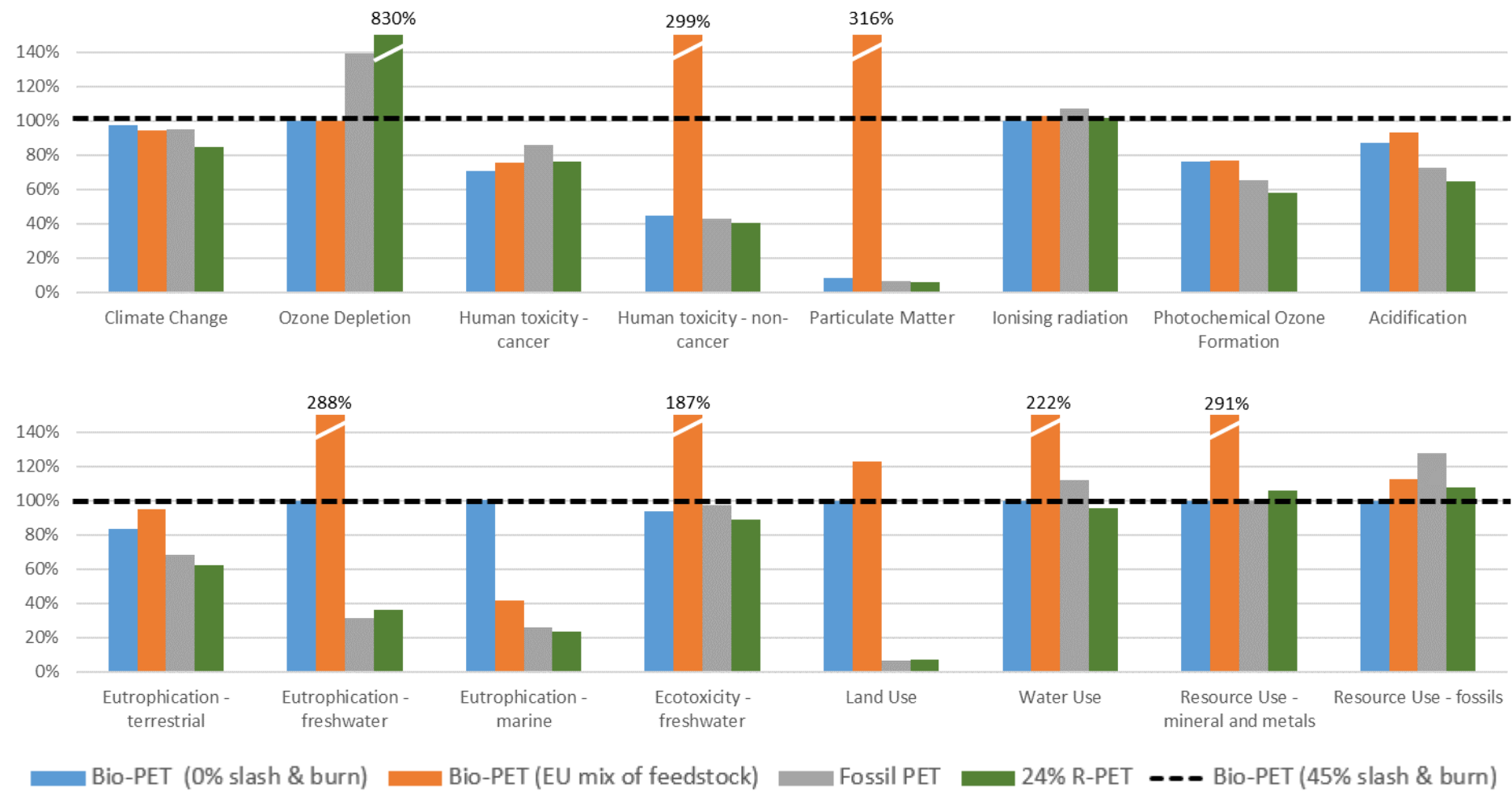
Completely phasing out the sugarcane pre-harvest burning practice decreases the impact of bio-based PET bottles in six impact categories, while no or irrelevant changes takes place in the remaining ones. The largest reductions are observed in Particulate Matter (-92%), Human Toxicity – non-cancer (-55%), Human Toxicity – cancer (-29%), and Photochemical Ozone Formation (-24%). In spite of these reductions, no substantial changes occur in the comparison with the reference scenario (i.e. fossil-based PET bottles), which is still preferable than, or comparable to bio-based PET bottles in the majority of the impact categories. However, compared to the base case (considering a 45% “slash and burn” rate), Bio-PET bottles are now preferable to fossil-based PET bottles in four categories instead of two (i.e. Ozone Depletion, Resource Use – fossils, Human Toxicity – cancer, and, to a lower extent, Water Use).

For Bio-HDPE bottles, the effects of totally removing the “slash and burn” practice are similar to those observed for Bio-PET ones, with the exception of a modest (and non-significant) increase of the impact in the category Resource Use – fossils (+5%). Also in

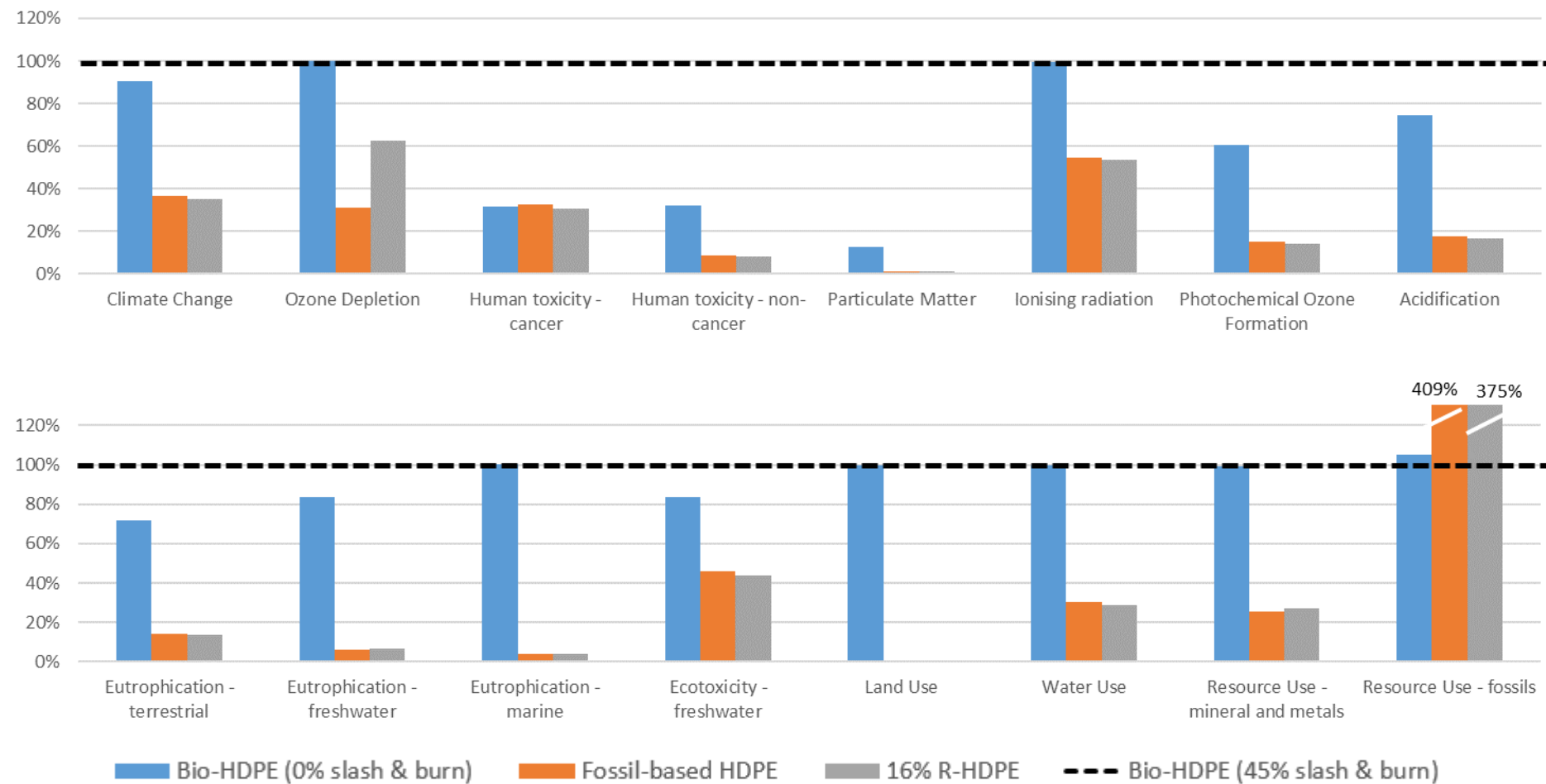
<sup>38</sup> The estimate is based on a specific consumption of starch for Bio-PET production equal to 0.85 kg starch/kg Bio-PET (IfBB, 2018), and a maize requirement of 0.948 kg maize grain / kg starch (according to maize starch production datasets in the Agrifootprint database).

1 this case, the achieved savings do not allow to cover the huge impact difference  
2 compared to fossil-based HDPE bottles, which remains the preferable alternative in all  
3 categories except for Resource Use - fossils (where Bio-HDPE bottles were already  
4 favoured), and Human Toxicity – cancer (where both alternatives are now comparable).

5 Replacing Brazilian sugarcane with the European mix of bioethanol crops increases the  
6 impacts of bio-based PET bottles in eight impact categories, with most of them showing  
7 an even important increase (in the range of 122%-216%). Only in three categories the  
8 impact is decreased (Photochemical Ozone Formation, Human Toxicity – cancer and  
9 Eutrophication – marine), while for the remaining categories the observed variations are  
10 irrelevant (or no changes occur). The reasons for this generally worsened picture could  
11 not be explored further, due to the use of aggregated datasets. However, one  
12 explanation may be the higher net energy demand of the processes required for the  
13 conversion of starch-based crops (maize, wheat) to bioethanol (compared to sugarcane),  
14 and the higher share of such crops in the applied mix compared to sugar beet (which  
15 requires a less energy intensive processing). This aspect is also reflected in available life  
16 cycle inventory datasets for bioethanol production (e.g. from the ecoinvent database),  
17 which show a larger impact of maize-based Ethanol compared to the use of sugarcane in  
18 many impact categories. On the other hand, Ethanol from sugar beet provides lower  
19 impacts in many categories compared to sugarcane, but its share in the mix is lower  
20 (21%). Differences in cultivation practices applied in Europe to the alternative bioethanol  
21 feedstock sources considered in this analysis may also play a role. In light of these  
22 results, bio-based PET bottles still show a worsened performance compared to fossil-  
23 based PET bottles in the majority of the impact categories, both alternatives being  
24 comparable in the remaining ones. An impact decrease is only observed for Ozone  
25 Depletion (-28%).



**Figure 4.22.** Results of the sensitivity analysis on the sugarcane “slash and burn” rate and on feedstock sourcing for bio-based PET beverage bottles. Scenario impacts are expressed as a percentage of the impacts of the base case of the affected scenario (i.e. bio-based PET bottles from Brazilian sugarcane with a 45% “slash and burn” rate), which is set as a baseline. Note that in some impact categories a part of the results is out of scale and is curtailed.



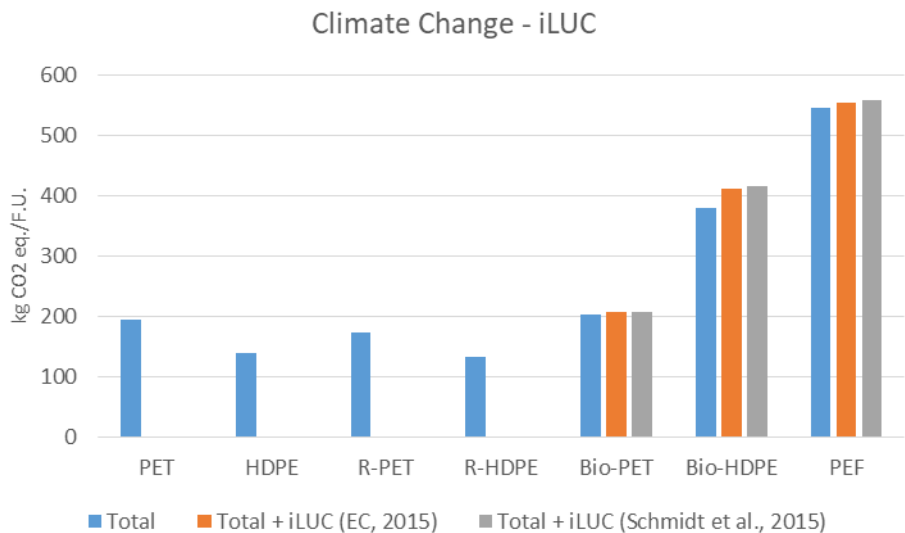
**Figure 4.23.** Results of the sensitivity analysis on the sugarcane “slash and burn” rate and on feedstock sourcing for bio-based HDPE beverage bottles. Scenario impacts are expressed as a percentage of the impacts of the base case of the affected scenario (i.e. bio-based HDPE bottles from Brazilian sugarcane with a 45% “slash and burn” rate), which is set as a baseline. Note that in some impact categories a part of the results is out of scale and is curtailed.



**4.7.8.3 iLUC model and factors**

This sensitivity analysis explores the use of an alternative model (i.e. Schmidt et al., 2015) to quantify the iLUC contribution to the Climate Change impact of bio-based bottles (i.e. Bio-PET, Bio-HDPE and PEF bottles). The iLUC factors calculated through the mentioned model were applied in place of the GHG emission factors from EC (2015) applied as a base case (Table 4.15). The results are displayed in Figure 4.24, limited to the Climate Change category, which is the only one affected by this sensitivity analysis.

Overall, the application of alternative iLUC factors only marginally affects the results. The highest impact increase observed when applying the (2015) factors was equal to 8.4% (sugarcane-based Bio-HDPE bottles), while applying the model from Schmidt et al. (2015) leads to a comparable increase of 9.1% (corresponding to a nearly 8% increase in relative terms). A similar increase is also observed for Bio-PET bottles (from 2.2% to 2.4%), these being still based on sugarcane as a feedstock (although limited to 30% of the polymer). For PEF bottles, the iLUC contribution was the lowest observed in base case results (1.4%), but it is the one showing the highest increase when the model from Schmidt et al. (2015) is applied, reaching 2.3%. However, both results are still comparable. In conclusion, the results of this case study can be considered reasonably robust with respect to the estimated iLUC contribution (which is modest or negligible), and the comparison among the affected beverage bottles scenarios and the respective fossil-based scenarios of reference is not affected by the application of alternative factors.



**Figure 4.24.** Results of the sensitivity analysis on the applied model to quantify the iLUC contribution to the total Climate Change impact of bio-based beverage bottles.

## 5 Case study 2: Flexible food packaging film

This case study assesses flexible packaging film for food products such as vegetables, dairy products or bakery products. The focus is on generic food packaging film with a fixed extension and a variable thickness to ensure comparable mechanical and barrier performances with the use of different materials. Therefore, the study does not address the use of plastic film for packing a specific food item (e.g. salad or bread) with a package of a defined size and capacity (e.g. bags containing 1 kg of product). This choice allows to keep a broader scope, as well as to explore a wider range of materials and feedstock sources within a single case study. It also simplifies some aspects of scenario setting and data collection, by avoiding the need to identify a representative average packaging for the product, and to define its features in each analysed material scenario. For similar reasons, mono-layer and mono-material films are considered in this case study, although multi-layer films are also commercially important (while not being dominant) in food packaging applications (McKeen, 2013). All these aspects are further discussed in the rest of this section.

### 5.1 Assessed scenarios

The different polymers and feedstock sources considered in the scenarios assessed for food packaging film in this case study are summarised in Table 5.1, which also specifies the End of Life options applied in each scenario. Conventional, fossil-based polymers considered as a reference for packaging film manufacturing are Polypropylene (PP; Scenario 1) and Low-density Polyethylene (LDPE; Scenario 2), which are the most commonly used materials for flexible food packaging applications (especially for fruit and vegetable products, bakery goods, snacks etc.; McKeen, 2013). Although bi-axially oriented PP (BOPP) is preferred in many packaging applications (e.g. for modified atmosphere packaging; Mangaraj et al., 2009), conventional non-oriented PP was considered here, to avoid any potentially biased comparison with the other investigated alternatives. BOPP is indeed characterised by improved mechanical and barrier properties (e.g. tensile strength and oxygen permeability) compared to non-oriented PP, and would thus be favoured in the performance-based comparison performed in this case study.

A fully bio-based, “drop-in” alternative to LDPE (i.e. Bio-LDPE) was assessed in Scenario 3, considering that the main precursor to bio-Ethylene production (i.e. bioethanol) is derived from Brazilian sugarcane (which is the most widely used feedstock at present for bioethanol-derived biopolymers). The use of alternative crops grown in Europe as a possible source of feedstock was explored as a sensitivity analysis (Section 5.7.7.2). Bio-based PP was not investigated, since the effects of feedstock substitution for a same conventional polymer are already explored for the case of LDPE, and to keep a reasonable and balanced number of reference and alternative scenarios to be assessed.

The use of three biodegradable alternative polymers (totally or partially bio-based) was also assessed for packaging film production (Scenarios 4-6). These include Polylactic Acid (PLA, totally bio-based, Scenario 4)<sup>39</sup>, a blend of Thermoplastic Starch (TPS) and PBAT (partially bio-based, Scenario 5)<sup>40</sup>, and a PLA/PBAT<sup>41</sup> blend (partially bio-based, Scenario 6)<sup>42</sup>. These materials (especially the two polymer blends) are characterised by an oxygen transmission rate similar to that of (oriented) PP, and hence represent a possible alternative to the use of this polymer in food packaging applications (e.g. for fresh-(cut) vegetable packaging). On the other hand, the water vapour transmission rate of such biodegradable materials is normally higher compared to PP (and LDPE as well), and this may be beneficial for products with high respiration rates.

<sup>39</sup> Bi-axially oriented PLA is specifically considered, according to the features of neat PLA packaging films most commonly found on the market.

<sup>40</sup> Referred to as Starch blend or Starch-based polymer (or film) in the following.

<sup>41</sup> PBAT: Polybutylene Adipate co-Terephthalate.

<sup>42</sup> Referred to as PLA blend or PLA-based polymer (or film) in the following.

For PLA production, US maize was considered as a feedstock, since nearly 90% of PLA currently consumed in Europe comes from the US (Eurostat, 2019a), where maize is used as a source of starch-derived sugar (dextrose) by the largest producer in the Country (Vink and Davies, 2015). However, the use of European maize was investigated as a sensitivity analysis, along with PLA production taking place in the EU (Section 5.7.7.3). Starch used for TPS production was derived from the current EU-average mix of starch crops, being one of the major producers of starch-based polymers located in Europe. The mix includes maize (47% on starch basis), wheat (40%) and potatoes (13%) (Starch Europe, 2019). PBAT used as a blending copolymer with either TPS or PLA is based on fossil sources, and no agricultural feedstock is hence needed in this case.

Finally, the use of a potential alternative feedstock for PP production, i.e. CO<sub>2</sub> captured from point emission sources, was explored in Scenario 7, representing an example of a Carbon Capture and Utilisation (CCU) pathway applied to the production of plastic polymers. CO<sub>2</sub> captured from coal-fired power plants was considered as a base case, these currently being the most abundant stationary source of CO<sub>2</sub> in Europe (Von der Assen et al., 2015) and worldwide (IPCC, 2005). The use of an alternative, more concentrate (but less abundant) source was explored as a sensitivity analysis, where CO<sub>2</sub> was derived from Ammonia production plants (Section 5.7.7.5). In the assessed scenarios, captured CO<sub>2</sub> is firstly converted to Methanol through hydrogenation, and then to Propylene via the Methanol-to-olefin route. As a base case, Hydrogen required for Methanol synthesis (along with captured CO<sub>2</sub>) was derived from the average mix of the main available production technologies, so as to reflect the current situation at the European level. Production pathways included in the mix were steam reforming of natural gas (48%), steam cracking of naphtha and other hydrocarbons (30%), coal gasification (18%) and electrolytic processes (4%), based on available data at the global level (OECD-IEA, 2015; Santhanam et al., 2017). However, one of the main drivers for investigating CO<sub>2</sub>-based pathways is the potential larger availability of “low-carbon” Hydrogen in the future (e.g. from water electrolysis or, to a lower extent and only under certain conditions, Methane cracking). This possibility was thus explored in a sensitivity analysis, considering Hydrogen produced from water electrolysis based on renewable energy as an input (Section 5.7.7.5).

Due to the innovative nature of the CO<sub>2</sub>-based route to PP production, life cycle inventory data applied in the modelling of relevant upstream activities (i.e. Methanol synthesis and conversion to Propylene via the Methanol-to-olefin route) were mostly derived from process simulation and modelling, rather than real production facilities. While possible process optimisation strategies are taken into account in the generation of such data (e.g. energy integration), they are not representative of large-scale industrial production (in contrast to data available for fossil-based PP considered as a reference) and do not account for possible (efficiency) improvement due to upscaling and further process optimisation. Therefore, a fully appropriate and consistent comparison between these two alternatives could not be ensured and the respective results need to be interpreted with more caution.

Regarding End of Life, all currently viable treatment and disposal options for packaging film made of each specific material were considered. For films made of materials not yet widely established on the market (e.g. biodegradable polymers), potentially viable options as of today were considered, taking into account relevant material properties (e.g. biodegradability), and the options currently applied to film made of the material(s) they intend to replace. Therefore, for film made of conventional, non-biodegradable polymers (i.e. PP and LDPE, from any feedstock) viable options that were considered include mechanical recycling, incineration and landfilling. For film made of biodegradable polymers (i.e. PLA, Starch/PBAT and PLA/PBAT) biological treatment options were also considered to be viable (i.e. composting and anaerobic digestion). Not that, the latter are not applicable to Bio-LDPE, which despite being bio-based is not biodegradable, having the same characteristics of its fossil-based counterpart (“drop-in” solution).

1 As a base case, the impacts of each packaging film scenario were assessed with  
2 reference to an EU-average End of Life scenario including all the options currently applied  
3 (or which would be potentially applied) at the EU level to film made of the specific  
4 material. These include all the viable options reported above, which were combined as  
5 described in Section 5.4.5.1 on End of Life modelling. In addition, scenario impacts were  
6 calculated by individually considering the application of each currently viable (or  
7 potentially viable) End of Life option.

8

**Table 5.1.** LCA scenarios assessed for the food packaging film case study.

Scenario	Polymer	Monomer or Co-polymer	Feedstock	End of Life options <sup>(1)</sup>
1 - Conventional polymer 1	PP	Propylene	Fossil-based (crude oil/natural gas)	Recycling Incineration Landfilling
2 - Conventional polymer 2	LDPE	Ethylene	Fossil-based (crude oil/natural gas)	Recycling Incineration Landfilling
3 - Alternative polymer 1	Bio-LDPE	Bio-Ethylene	Sugarcane (BR)	Recycling Incineration Landfilling
4 - Alternative polymer 2	PLA (Bi-axially oriented)	Lactic Acid	Maize (US)	Recycling Composting Anaerobic digestion Incineration Landfilling
5 - Alternative polymer 3	TPS/PBAT blend <sup>(2)</sup>	TPS 1,4-Butanediol Adipic acid PTA <sup>(3)</sup>	EU mix of starch crops <sup>(4)</sup> Crude oil/natural gas	Recycling Composting Anaerobic digestion Incineration Landfilling
6 - Alternative polymer 4	PLA/PBAT blend <sup>(5)</sup>	Lactic acid 1,4-Butanediol Adipic acid PTA <sup>(3)</sup>	Maize (US) Crude oil/natural gas	Recycling Composting Anaerobic digestion Incineration Landfilling
7 - Alternative polymer 5	CO <sub>2</sub> -based PP	CO <sub>2</sub> -based Propylene	CO <sub>2</sub> (coal fired power plants) H <sub>2</sub> (average prod. mix) <sup>(6)</sup>	Recycling Incineration Landfilling

<sup>(1)</sup> The impacts of scenarios were individually assessed for each listed End of Life option, as well as for a combination of such options reflecting as far as possible the current (real or potential) average situation at the EU level.

<sup>(2)</sup> A 40% Starch and 60% PBAT blend is considered, according to the information received from stakeholders' consultation, reporting that "renewability" of starch-based polymers used in packaging applications can reach 40%. The use of an increased starch content was explored as a sensitivity analysis.

<sup>(3)</sup> PTA: Purified Terephthalic Acid

<sup>(4)</sup> The mix includes Maize (47%), Wheat (40%), Potatoes (13%), in terms of starch product equivalents (Starch Europe, 2019).

<sup>(5)</sup> The blend includes 70% PLA and 30% PBAT, according to examples reported in Choi et al. (2018) and Pivsa-Art et al. (2013).

<sup>(6)</sup> Steam reforming of natural gas (48%), steam cracking of naphtha and other hydrocarbons (30%), coal gasification (18%), electrolytic processes (4%) (OECD-IEA, 2015; Santhanam et al., 2017).

## 5.2 Functional Unit and reference flow

The main function of the studied article would be, once converted into a specific packaging item with a defined size and capacity, the delivery of packaged food to final consumers. In this case, the functional unit could be defined, for instance, as "delivering 1 kg of a specific food item for actual consumption by the final consumer, ensuring a

*comparable product (shelf) life*". However, given the broader focus of this case study on generic food packaging film, rather than on a defined packaging article for a specific food product, a more general, extension-based functional unit was selected, i.e. *"providing 100 m<sup>2</sup> of flexible packaging to a generic food product in the EU, without breaking during transport and handling under a minimum specified tensile load, and ensuring a comparable shelf life for the packaged product"* (Table 5.2).

In this study, it was assumed that the shelf life is uniquely determined by the oxygen permeability of the film, reflecting the Oxygen Transmission Rate (OTR) of the respective material and the thickness of the film itself. Conversely, the ability of the film of not breaking under a minimum specified tensile load (i.e. one of the most important stresses that film may undergo) mainly depends on the tensile strength of such materials (beyond the final thickness of the film).

**Table 5.2.** Definition of the functional unit for food packaging film LCA scenarios.

Aspect	Description
"What" (function(s) or service(s) provided)	Providing flexible packaging to a generic food product
"How much" (extent of the function(s) or service(s))	100 m <sup>2</sup> of flexible packaging
"How well" (expected level of quality)	Without breaking during transport and handling under a minimum specified tensile load, and ensuring a comparable shelf life for the packaged product
"How long" (duration/lifetime of the function or service)	Over the product (shelf) life
"Where" (location/geography of the service)	In the EU

The different compared materials have different technical properties (e.g. mechanical and permeability properties) for a given film thickness, affecting their ability to achieve a defined performance/functionality requirement (or design constraint) such as a given tensile strength or oxygen barrier. These differences in technical properties may thus ultimately affect the mass of material needed to fulfil the functional unit (i.e. the reference flow), as far as the respective performance requirements can be met by adjusting the thickness of the film over its defined extension (i.e. 100 m<sup>2</sup>). Depending on the specific performance/functionality requirement that is considered (or design constraint that needs to be prioritised), a different film thickness, and hence material mass, is generally needed. Therefore, a limiting requirement or constraint can be identified in the one implying the highest thickness (and mass) of material. This ensures that also the remaining requirements are met, provided that these do not counteract each other (e.g. if having a thicker film does not imply a worsened performance towards any other requirement).

For illustrative purposes, in this case study it was assumed that a single performance requirement could be prioritised at the time (regardless of whether the other requirements are met or not), to independently evaluate how the comparative performances of the assessed scenarios are affected by each specific material property. As a base case, the need for the film of not breaking during transport and handling under a minimum specified tensile load (and hence of ensuring a minimum tensile strength<sup>43</sup>) was assumed as the performance requirement to be prioritised (or the limiting design constraint). In particular, the average tensile strength of non-oriented Polypropylene film was considered as the reference performance to be achieved (i.e. 37 N/mm<sup>2</sup>, according to Marangaj et al., 2009), assuming that this value can be considered representative of a

<sup>43</sup> Tensile strength represents the ability of a material to withstand a specified tensile load without breaking.

film with a typical median thickness of 30  $\mu\text{m}$ <sup>44</sup>. However, since the different compared materials may perform differently towards different technical properties, a sensitivity analysis was conducted to evaluate the effects of prioritising the need of ensuring a comparable shelf life (and hence oxygen permeability) as the alternative performance requirement to be fulfilled. In this respect, it is noted that other performance requirements may be considered relevant for the assessment, depending on the specific situation and the ultimate packaging application (e.g. ensuring no deformation under specified tensile loads).

No immediate deterministic relationships are available to relate the tensile strength (or other technical properties) of a given material, with the thickness of the film (and hence with the mass of polymer) needed to achieve a required (strength) performance. In addition, technical data sheets of several packaging films on the market reports no variations of tensile strength within the offered range of thicknesses (e.g. for BO-PP, BO-PLA and LDPE), while in the literature a few controversial evidence is found. For instance, tensile strength of blown LDPE film was found to be independent of thickness between 50  $\mu\text{m}$  and 400  $\mu\text{m}$ , while it increased for thinner films with a thickness of 15  $\mu\text{m}$  or lower (Rennert et al., 2013). Conversely, tensile strength of starch-based films increased with thickness from 300  $\mu\text{m}$  to 1 mm, while it decreased if thickness was further increased beyond 1.5 mm (Jansson and Thuvander, 2004).

Therefore, to calculate the reference flow (i.e. the amount of polymer required to fulfil the functional unit), accounting for differences in relevant technical properties of the compared materials, the concept of Material Substitution Factors (MSFs) was applied. MSFs express the ratio between the mass of a given material and that of a reference material, needed to perform identically a given technical function, and can be calculated on the basis of so-called Material Indices (MIs). MIs were introduced by Ashby (1999) to facilitate initial material selection in mechanical design, and aims to score the performance of a material with respect to a given (mechanical) function/performance (e.g. strength or stiffness), based on relevant material properties. These typically include the density of the material and technical properties relevant to the considered function (e.g. tensile strength or Young's modulus). Materials with the highest Material Index can perform the intended function with the minimum mass<sup>45</sup>. Therefore the reciprocal of the ratio between Material Indices of the reference material and of an alternative one, can be considered to represent the Material Substitution Factor, according to the following Equation:

$$\text{MSF} = m/m_{\text{ref}} = \text{MI}_{\text{ref}}/\text{MI}$$

where  $m$  and  $m_{\text{ref}}$  respectively represent the mass of the alternative and of the reference material needed to perform the intended function, while  $\text{MI}$  and  $\text{MI}_{\text{ref}}$  represent the Material Indices of the same materials, with respect to such function.

The Material Index reported by Ashby (1999) when strength is the priority function (or the limiting constraint) for structural components subject to tension (tie, with thickness as a free variable) is represented by the specific strength, i.e. the ratio between the tensile strength ( $\sigma_y$ ) and the density ( $\rho$ ) of the material (i.e.  $\text{MI} = \sigma_y/\rho$ ). This index was thus considered suitable to calculate MSFs for the alternative packaging film materials investigated in this case study, and the corresponding reference flow. The main calculations carried out for this purpose, and the values assumed for relevant material properties are summarised in Table 5.3. As discussed above, (non-oriented) Polypropylene was considered as the reference material for calculation purposes, assuming it is used in a film with a typical median thickness of 30  $\mu\text{m}$ , and accounting (where relevant) for technical properties associated with this thickness.

It is noted that the purpose here is to illustrate the concept of integrating relevant material properties in the calculation of the reference flow, for a more appropriate,

<sup>44</sup> Considering the typical range of thicknesses offered for (BO)-PP film currently available on the market.

<sup>45</sup> Note, however, that material indices do not express the mass of material needed to perform a given (mechanical) function, but only the performance of the material with respect to that function.

performance-based comparative assessment. In this perspective, the study does not aim at providing accurate estimates of the potential impacts of each packaging film material when a given technical performance needs to be ensured. Therefore, the calculation approach adopted in this study is considered suitable, despite it provides estimates of material requirements that may not be necessarily reflected in real products on the market.

**Table 5.3.** Calculation of the reference flow for food packaging film LCA scenarios when tensile strength is the prioritised performance requirement to be fulfilled (the calculation is based on material substitution factors).

Polymer	Density - $\rho$ (kg/m <sup>3</sup> )	Tensile strength - $\sigma_y$ (N/mm <sup>2</sup> )	MI ( $\sigma_y/\rho$ )	MSF (m/m <sub>ref</sub> )	Reference flow (kg/FU)	Thickness ( $\mu$ m) ( <sup>1</sup> )
PP (all types of feedstock)	900 ( <sup>2</sup> )	37 ( <sup>2</sup> )	41.1	1	2.70 ( <sup>3</sup> )	30
LDPE (all types of feedstock)	928 ( <sup>2</sup> )	19.5 ( <sup>2</sup> )	21.0	1.96	5.28	57
PLA (bi-axially oriented)	1,240 ( <sup>4</sup> )	83.5 ( <sup>4</sup> )	67.3	0.61	1.65	13
Starch/PBAT blend	1,220 ( <sup>5</sup> )	43 ( <sup>6</sup> )	35.2	1.17	3.15	26
PLA/PBAT blend	1,290 ( <sup>7</sup> )	34 ( <sup>8</sup> )	26.4	1.56	4.21	33

(<sup>1</sup>) The thickness of PP film (reference material) was assumed (and used to calculate the reference flow, see note 3). For films made of the other materials, the thickness was backward calculated for checking purposes, based on the estimated reference flow and the assumed film extension (100 m<sup>2</sup>). The estimated thickness values are realistic, although the relatively low value obtained for PLA film falls outside the typical range of thicknesses offered on the market for film made of this material (20-100  $\mu$ m).

(<sup>2</sup>) Average of the range (extreme values) reported in the review by Mangaraj et al. (2009). Values of tensile strength are not reported as a function of thickness.

(<sup>3</sup>) For PP film (reference material), the reference flow is calculated assuming a film thickness of 30  $\mu$ m, representing the typical median value of the thickness range offered for some (BO)-PP films available on the market. This thickness could not be specifically related to the assumed average value of tensile strength, and it is noted as a partial inconsistency.

(<sup>4</sup>) Average of the values from the technical specifications of two BO-PLA packaging films available on the market. Tensile strength is not reported as a function of the offered range of thickness (i.e. a single value is reported).

(<sup>5</sup>) Based on the technical specifications of a starch-based packaging film.

(<sup>6</sup>) Based on the experimental results from Briassoulis and Giannoulis (2018), which refer to a 25  $\mu$ m Starch-based film.

(<sup>7</sup>) Estimated based on linear regression on density values of a 100% PLA film (1240 kg/m<sup>3</sup>) and of a PLA/PBAT film with a 34% PLA content (1340 kg/m<sup>3</sup>).

(<sup>8</sup>) Based on the experimental results from Briassoulis and Giannoulis (2018), which refer to a 30  $\mu$ m PLA/PBAT film.

### 5.3 System boundary

In all scenarios, the system boundary was set in order to cover the most relevant stages and processes of the full product life cycle (cradle-to-grave perspective), as described below and depicted in Figures 5.1 to 5.7:

- *Feedstock Supply*<sup>46</sup> – covering extraction, transport and possible refining of crude oil and natural gas (fossil-based polymers), crop cultivation (bio-based polymers), CO<sub>2</sub> capture and H<sub>2</sub> production (CO<sub>2</sub>-based PP), as well as transport of these feedstock sources to downstream conversion or utilisation processes (e.g. naphtha

<sup>46</sup> Corresponding to the default stage of "Raw Material Acquisition and Pre-Processing" specified in the Method and in the PEF framework.



- 1 cracking, sugarcane fermentation, wet milling of starch crops, CO<sub>2</sub>-based  
2 Methanol production);
- 3 • *Polymer Production*<sup>47</sup> – covering all the activities associated with the conversion of  
4 feedstock materials into the relevant monomer(s) and final polymer, including any  
5 transport among these activities and final transport of polymer granulate to  
6 downstream manufacturing processes;
  - 7 • *Article Production*<sup>48</sup> – including food packaging film manufacturing through blown  
8 film extrusion of polymer granulates and transport of film to the food packing site;
  - 9 • *Distribution* – including transport of film (in the form or as part of a final  
10 packaging item) from the food packing site to retailers and from these to final  
11 consumers<sup>49</sup>;
  - 12 • *End of Life* – covering collection, transport, recycling, biological treatment,  
13 incineration, or disposal of packaging film after use, including any avoided  
14 processes from virgin material or energy substitution.

15 A different nomenclature was applied for some life cycle stages compared to the default  
16 nomenclature specified in the Method (and in the PEF framework) to make it more  
17 relevant for the investigated supply chains and the project scope. Moreover, the default  
18 “Raw Material Acquisition and Pre-processing” stage was further split into two separate  
19 sub-stages (i.e. Feedstock Supply and Polymer Production), to allow disaggregating the  
20 impacts of feedstock supply from downstream conversion processes, and hence to better  
21 appreciate any differences among the use of different feedstock sources (in line with the  
22 project goal).

23 The food packing process (e.g. filling and sealing), the Use Stage, and the life cycle of  
24 any other packaging components (e.g. food trays where film may be used for lidding or  
25 wrapping) were excluded from the assessment. This is because this study focuses on the  
26 packaging film life cycle (and on the function(s) provided by this) and not on the life  
27 cycle and function(s) of a specific packaged food or of a specific packaging item (totally  
28 or partially relying on plastic film). Moreover, the burdens of the mentioned activities and  
29 stages can be considered identical in all the investigated scenarios, since the use of  
30 different film materials or feedstock sources do not usually affect the burdens of the  
31 packing process (e.g. energy consumption), or of the Use Stage (e.g. refrigeration  
32 requirements or other relevant aspects)<sup>50</sup>, nor the type of any additional packaging  
33 components<sup>51</sup>. Therefore, such activities and components can be excluded from a  
34 comparative assessment, as the one performed in this study, and their exclusion do not  
35 affect the outcome of the comparison among the different scenarios.

36 It has to be noted, however, that the mentioned exclusions may weaken the assessment  
37 if they are not adequately justified (e.g. if relevant differences exist among compared  
38 alternatives, which are not taken into account). Therefore, the Use Stage, food packing  
39 activities (and the life cycle of any additional packaging components) shall always be  
40 taken in full consideration for possible inclusion in a comparative LCA study on food  
41 packaging film (or on specific food packaging items), paying particular attention to any

<sup>47</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

<sup>48</sup> Corresponding to the default stage of “Manufacturing” specified in the Method and in the PEF framework.

<sup>49</sup> While filled packages obtained from flexible film are transported in reality (with the respective product content), transport of packaged food was excluded in this case study, due to the exclusive focus on the packaging film life cycle, and not on the life cycle of a specific packaged food product.

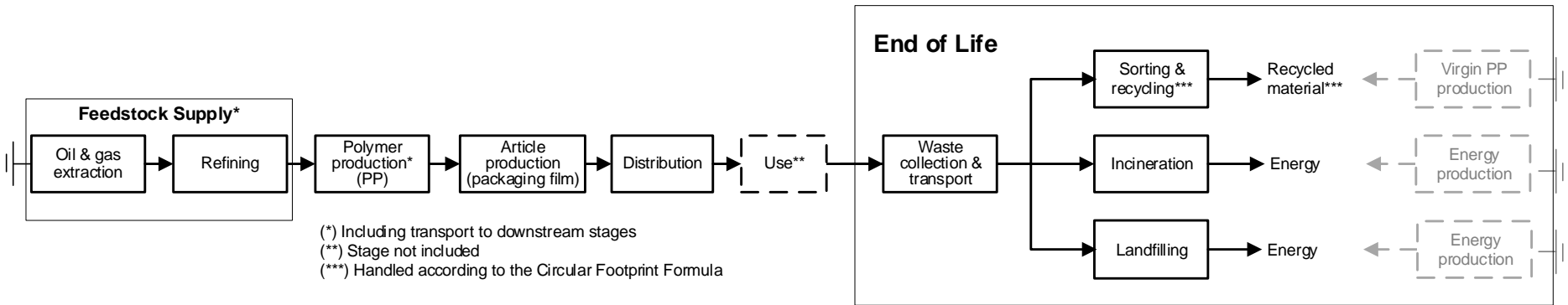
<sup>50</sup> This is true as long as different packaging materials do not imply significantly different product (shelf) life, which may affect the storage duration (with the respective energy requirements, in case refrigeration is needed) as well as the potential generation of food waste at consumer level. In this situation, the Use Stage should be taken into account, although changes in food waste generation due to different shelf lives are hardly quantifiable. However, in this case study, the requirement to ensure comparable mechanical and barrier performances expressed in the functional unit, imply that a similar product (shelf) life is achieved by all the compared alternatives, without affecting Use Stage burdens.

<sup>51</sup> Unless a packaging item with specific properties (e.g. biodegradability) is to be produced, so that materials with suitable characteristics would need to be used.

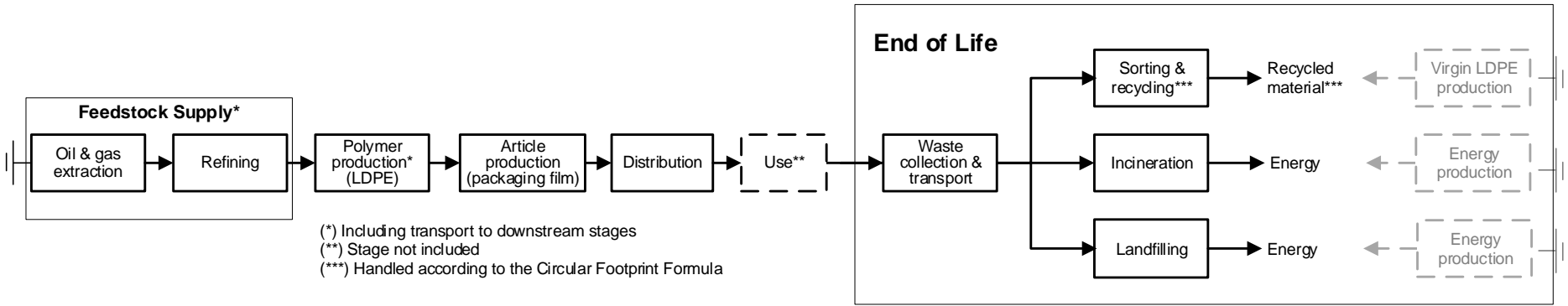
1 differences among the compared alternatives. Moreover, such stages shall always be  
2 included in LCA studies of packaged food products, in line with system boundary  
3 requirements in the Method and in the PEF framework.

4 Finally, it has to be noted that additives were not included in the assessment, due to the  
5 lack of complete and consistent data on the use of additives in the production of food  
6 packaging film, of the examined polymers, and of plastics in general, as well as on their  
7 release and fate over the product life cycle. One exception is represented by additives  
8 (mainly plasticisers) used in Thermoplastic Starch production, which represent a non-  
9 negligible and essential portion of the final polymer (i.e. 25%) and are required to ensure  
10 suitable material properties and processability. Moreover, production of additives used in  
11 starch-based polymers can account for a non-negligible portion of the respective cradle-  
12 to-gate Climate Change impact and energy demand, reaching 46% for polymer grades  
13 including larger shares of additives in the range of 30% (Broeren et al., 2017). Leaving  
14 this exception apart, the exclusion of additives is acknowledged as a limitation of this  
15 study, since additives can also be relevant at the End of Life stage, where they can be  
16 released, as such or after degradation/conversion into different compound(s), in the  
17 environment.

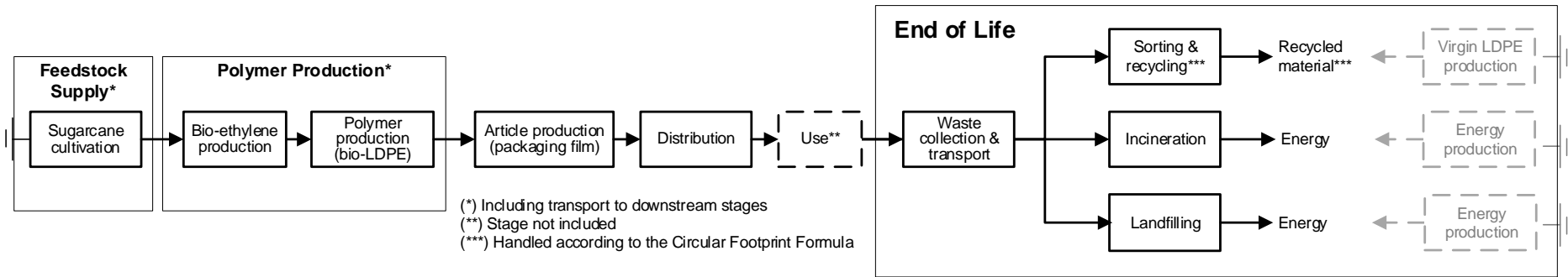
18



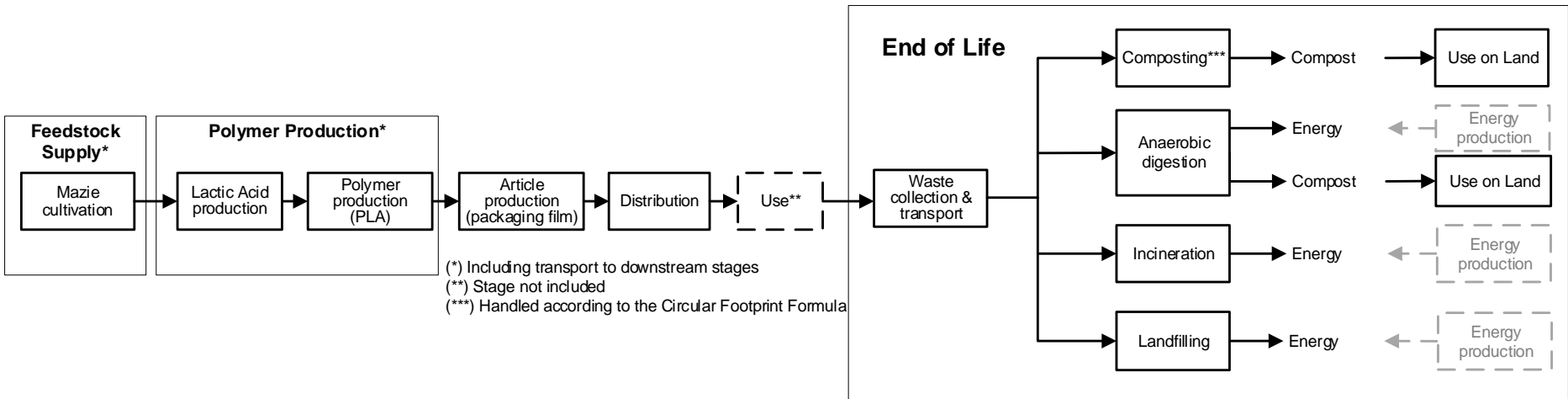
**Figure 5.1.** System boundary for fossil-based PP food packaging film (Scenario 1).



**Figure 5.2.** System boundary for fossil-based LDPE food packaging film (Scenario 2).



**Figure 5.3.** System boundary for bio-based LDPE food packaging film (Scenario 3).



**Figure 5.4.** System boundary for PLA food packaging film (Scenario 4).

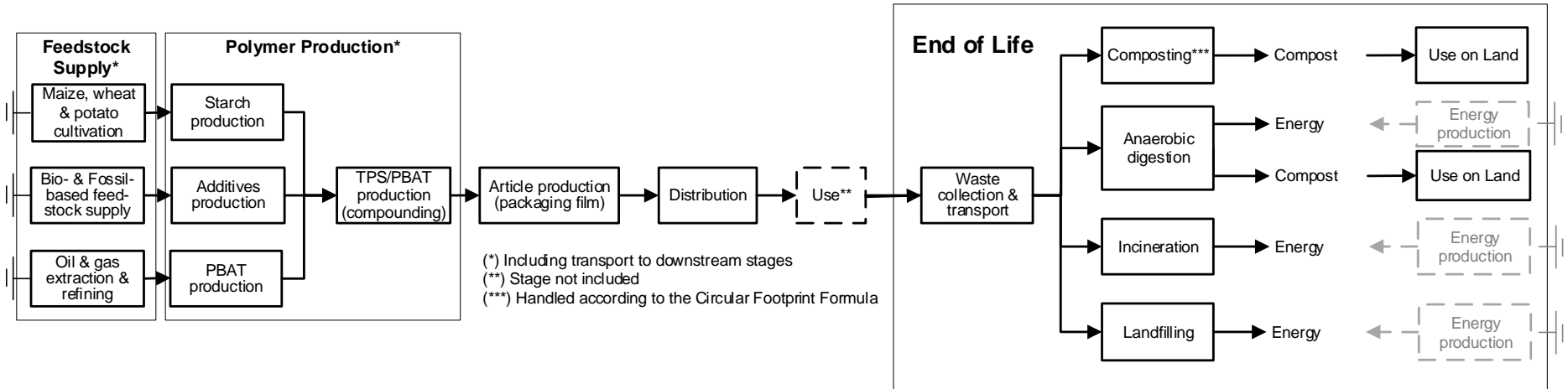


Figure 5.5. System boundary for Starch-based food packaging film (Scenario 5).

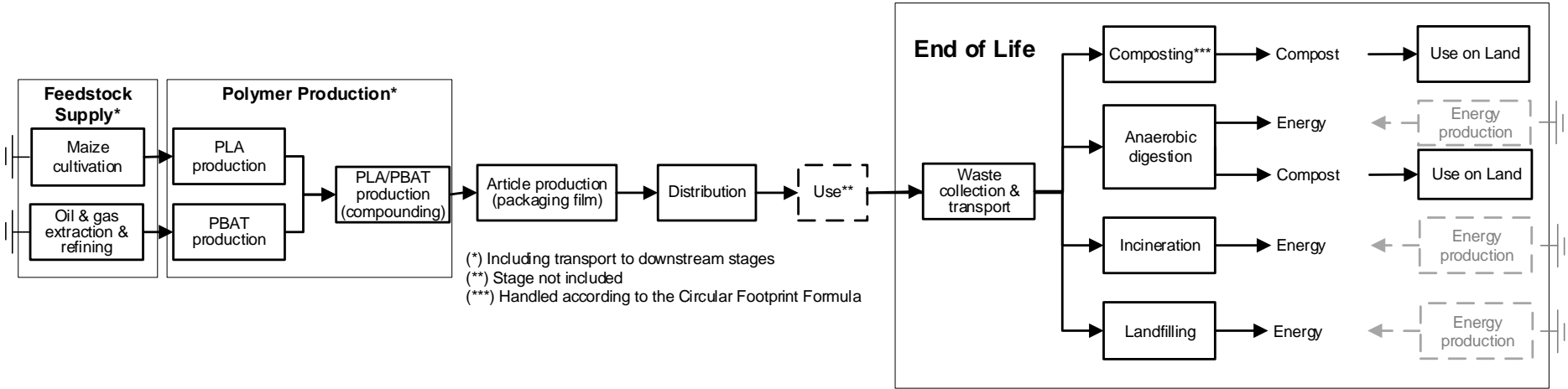
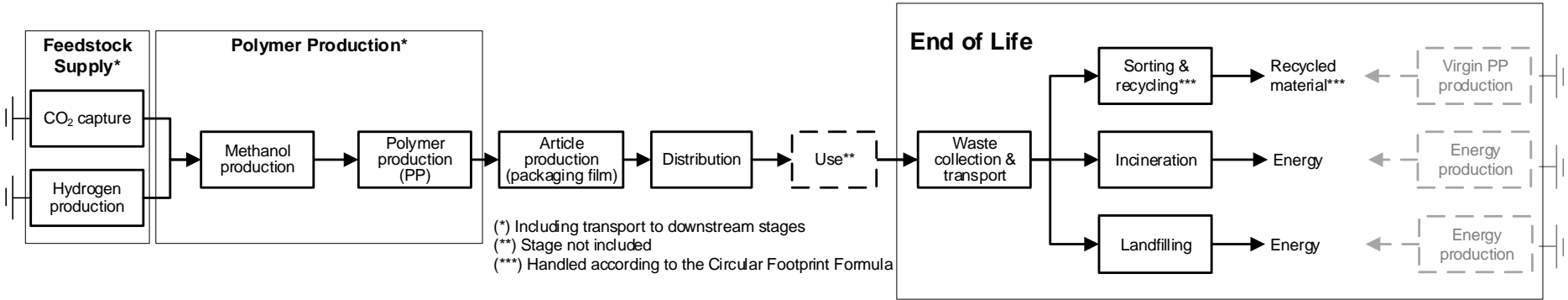


Figure 5.6. System boundary for PLA-based food packaging film (Scenario 6).



**Figure 5.7.** System boundary for CO<sub>2</sub>-based PP food packaging film (Scenario 7).

## **5.4 Life Cycle Inventory**

This section describes the overall approach for developing the Life Cycle Inventory of the analysed scenarios, along with the related assumptions and data sources. The description is separated by major lifecycle stages, in the following sub-sections (6.4.1 – 6.4.6). The list of processes, related data sources, and main modelling details are provided in Tables B.2.1 to B.2.7 in Annex B.2.

### **5.4.1 Feedstock Supply Stage**

#### **5.4.1.1 Fossil-based polymers**

For fossil-based polymers (PP and LDPE), the stage of Feedstock Supply includes the processes of crude-oil and natural gas extraction and transport, naphtha production in crude oil refineries, and its subsequent transport to downstream conversion processes (i.e. naphtha cracking). The modelling of these activities was carried out as described in Section 4.4.1.1, where the reader is referred to for further details on data sources, covered sub-processes and related environmental burdens, main modelling choices (e.g. allocation) and any adjustments performed in the implementation of the applied datasets.

For PBAT used as a copolymer in PLA-based and Starch-based packaging film, the stage of Feedstock Supply could not be separately modelled from downstream conversion and polymerisation processes, in the absence of disaggregated datasets related to 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 is hence entirely discussed in Section 5.4.2.2, focusing on Polymer Production.

#### **5.4.1.2 Bio-based polymers**

For bio-based polymers or copolymers (Bio-LDPE, PLA and Thermoplastic Starch), 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.

Growing of Brazilian sugarcane (used as a feedstock for bio-Ethylene) was modelled through aggregated datasets from the GaBi database. Used in combination, these datasets depict a situation where 45% of sugarcane is manually harvested via the "slash and burn" practice, i.e. sugarcane residues (tops and leaves) are burned on standing plants before harvesting. This share was considered representative of the current situation. However, the slash and burn 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 phasing out of this practice (see Section 5.7.7.2). Transport of harvested sugarcane to further processing in sugarcane mills for Ethanol 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 US Maize for PLA production, and of the European starch crops (maize, wheat, potatoes) used as a feedstock for Thermoplastic Starch, was modelled through available EF-compliant datasets. These are based on the approach used to develop agricultural inventories in the Agri-footprint database, relying on 5-year average yield data from FAOStat (2010-2014). Activities covered in the datasets are those of seeding and seed production, fertilizer and pesticide production and application, capital goods, as well as energy use and transport for field management practices. 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 US or EU diesel mix, depending on the country of production).

### 5.4.1.3 CO<sub>2</sub>-based PP

Activities related to Feedstock Supply for CO<sub>2</sub>-based PP production include capture, purification and liquefaction of CO<sub>2</sub> from flue gas of coal-based power plants (or the sole purification and liquefaction of the almost pure gaseous CO<sub>2</sub> output from Ammonia production), Hydrogen production<sup>52</sup>, and transport of these raw materials to downstream conversion (utilisation) facilities in a liquefied form via pipelines (assumed to be the most suitable transport form for industrial use on larger scales).

As a base case, CO<sub>2</sub> was assumed to be a waste of the considered CO<sub>2</sub> source (coal-fired power plants or ammonia production plants), according to the discussion reported in Section 4.4.4 of the Method (Report I), and especially considering that raw gaseous CO<sub>2</sub> has no economic value at the point of its arising. This assumption is also in line with the discussion by Giegrich et al. (2018), who argued that CO<sub>2</sub> can be considered a waste if it has no value before capturing, or if its supply is much larger than its demand in the context of the CO<sub>2</sub> emitter (as it is currently the case for power plants and many other CO<sub>2</sub> sources). In this perspective, the subsequent processes of capture<sup>53</sup>, compression, transport and utilisation constitute the components of a recycling chain aimed at converting waste CO<sub>2</sub> into a useful CO<sub>2</sub>-based product (i.e. Methanol and, later, Propylene), ultimately replacing an equivalent product from primary resources. Therefore, the Circular Footprint Formula was applied, as in any other recycling situation, to model the environmental burdens of feedstock CO<sub>2</sub> and of the resulting CO<sub>2</sub>-based product (as better discussed in Section 4.4.4 of the Method). Alternative approaches were explored as a sensitivity analysis (Section 5.7.7.6), applying also a wider system perspective where the CO<sub>2</sub>-based product (i.e. CO<sub>2</sub>-based Propylene) is considered, along with the main product of the CO<sub>2</sub> source (i.e. coal-based electricity) a co-product of the overall Carbon Capture and Utilisation (CCU) system.

The application of the CFF implied that a share of the burdens of the capture and utilisation (i.e. “recycling”) process-chain (including capture, purification, liquefaction, transport and utilisation) were assigned to the CO<sub>2</sub>-based product. Conversely, no burdens from any upstream activity related to supply and conversion of the CO<sub>2</sub>-providing fossil-based feedstock (e.g. coal extraction and combustion) were attributed to it, as these activities take place before waste CO<sub>2</sub> arising. On the other hand, the CO<sub>2</sub>-based product carries a share of the burdens associated with primary production of the conventional product it replaces, including supply of the respective primary feedstock. In principle, the most proximate point of substitution within the supply chain should be considered, which in the case of CO<sub>2</sub>-based PP could be identified at the level of Methanol production (with CO<sub>2</sub>-based Methanol replacing conventional Methanol). However, since the subsequent Propylene synthesis from Methanol relies on a completely different route compared to the one commonly applied for fossil-based Propylene production (i.e. naphtha cracking), it was considered more appropriate to assume substitution at the Propylene level (with CO<sub>2</sub>-based Propylene from the Methanol-to-olefin route replacing fossil-based Propylene from steam cracking). This assumption may anyway need to be reconsidered in the future, taking into account the actual penetration of the Methanol-to-olefins route as a practice for fossil-based Propylene production from other feedstock sources than crude oil (e.g. coal or natural gas). The share of the burdens of the two production pathways assigned to the CO<sub>2</sub>-based product depends on the value of the A factor, which should be defined by taking into account the relation between the supply

<sup>52</sup> Supply of fossil-based feedstock sources (natural gas, oil, coal) used for the majority of Hydrogen production could not be modelled separately from downstream conversion processes, due to the lack of disaggregated production datasets for these conversion pathways. All cradle-to-gate activities related to the supply of Hydrogen were thus modelled and accounted as a part of the stage of Feedstock Supply. This is considered consistent for this specific (CO<sub>2</sub>-based) production pathway, since the supply of the two “primary” building blocks (i.e. CO<sub>2</sub> and Hydrogen) for downstream conversion processes is addressed at the same level in the life cycle.

<sup>53</sup> Only if capture is purposefully carried out at the CO<sub>2</sub> source to enable its downstream utilisation. This is not the case, for instance, of Ammonia production plants, where CO<sub>2</sub> separation/extraction from the gaseous mixture of CO<sub>2</sub> and Hydrogen generated in the process is necessarily carried out as an integral part of the process itself to allow subsequent use of Hydrogen for Ammonia synthesis.



and demand of the “recycled” (in this case, CO<sub>2</sub>-based) product. Since CO<sub>2</sub>-based olefins are currently not available on the market, the evaluation was conducted assuming a “neutral” situation of equilibrium between supply and demand of CO<sub>2</sub>-based Propylene (i.e. A=0.5), which is expected to occur once a demand for CO<sub>2</sub>-based Propylene would be established, considering the current large availability of CO<sub>2</sub> sources. According to this choice, only 50% of the burdens from the processes of capture, purification, liquefaction and transport of waste CO<sub>2</sub> were assigned to the CO<sub>2</sub>-based product (i.e. Propylene). On the other hand, the latter carries 50% of the burdens associated with the supply of fossil-based feedstock required for the production of the replaced fossil-based Propylene (which has been modelled as already described in Section 5.4.1.1).

The CO<sub>2</sub> capture, purification and liquefaction processes were modelled based on the *ecoinvent* dataset “[RER] Carbon dioxide, liquid”, which refers to CO<sub>2</sub> extraction from (waste) gas streams of industrial production processes by means of a 15-20% Monoethanolamine (MEA) solution, followed by purification and liquefaction. Compared to the original dataset, the electricity consumption was updated with a more representative value (1.32 MJ/kg CO<sub>2</sub>), calculated as the average of the values specifically reported in Von der Assen et al. (2015) for CO<sub>2</sub> extraction from flue gases of coal-based power plants using MEA as a solvent. On the other hand, the default values applied in the dataset for capture efficiency, MEA consumption, water flows, and air emissions were considered representative also for extraction from flue gas of power plants (except for Methane emissions that were removed). Finally, background datasets related to energy supply (in this case only electricity) were replaced with EF datasets, while infrastructure processes (related to chemical factory construction) were removed, to improve reliability of LCIA results in the Ozone Depletion impact category.

For Hydrogen production from fossil-based sources (natural gas, oil-based hydrocarbons and coal) aggregated, cradle-to-gate inventory datasets from the GaBi or EF databases were separately applied in the modelling (as better detailed in Table B.2.7 in Annex B.2). The dataset related to Hydrogen production from steam reforming of natural gas refers to German background conditions, but considering the limited number and the type of background activities involved (natural gas supply, deionised water production and electricity substitution) it can be considered representative of average conditions in Central Europe, which is a reasonable approximation for this study. The same consideration applies to the dataset related to Hydrogen production from coal gasification, which in the absence of specific data for this pathway was approximated with production via steam reforming of heavy fuel oil. For Hydrogen from water electrolysis, a separate inventory was built by combining foreground inventory data reported by Hoppe et al. (2017, which in turn refer to BTS et al., 2014 and Hotellier et al., 2014) and background EF datasets for the corresponding process inputs. Consistently with the assumption performed in the base case scenario regarding Hydrogen supply, the process was fed with average electricity from the EU grid mix. The use of renewable energy was explored in a sensitivity analysis, assuming that Hydrogen is entirely supplied from water electrolysis as a “low-carbon” source of this element.

Transport of liquefied CO<sub>2</sub> to downstream industrial users via pipelines was assumed to take place along a distance of 300 km, in line with the “CO<sub>2</sub> deserts map” reported in Von der Assen et al. (2015) for CO<sub>2</sub> sources located in Europe. According to the latter, considering only CO<sub>2</sub> sources available today, the current European demand of 50 Mt CO<sub>2</sub>/year can be met by CO<sub>2</sub> sources located at a distance not larger than 300 km from hypothetical users in central Europe (and demanding up to 5 Mt CO<sub>2</sub>/year). The same distance assumed for CO<sub>2</sub> was then consistently applied also to transport of Hydrogen.

The burdens of onshore pipeline transport of both liquefied CO<sub>2</sub> and Hydrogen were approximated with those of long-distance transport of natural gas reported in the *ecoinvent* dataset “[DE] transport, pipeline, long distance, natural gas”. 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 datasets for electricity generation in the EU. Moreover, leakage of

natural gas and the related air emissions were replaced with leakage and emissions of CO<sub>2</sub> or Hydrogen, respectively.

### **5.4.2 Polymer Production Stage**

The Polymer Production stage covers the activities of feedstock processing into relevant intermediates and monomer(s), the polymerisation process, possible compounding of polymer blends, as well as any transport among these activities and final transport of polymer granulate to the packaging film manufacturing site. The following subsections (5.4.2.1 – 5.4.2.4) describe how these activities have been modelled in the present case study.

#### **5.4.2.1 Fossil-based polymers**

For conventional, fossil-based polymers (PP and LDPE), the whole process chain from feedstock processing to polymerisation, through the production of intermediates and monomers, was modelled by means of aggregated, gate-to-gate datasets provided by Thinkstep. Inputs includes combinations of crude oil, natural gas and naphtha, depending on the polymer. All conversion processes are assumed to take place in Europe, so that the datasets reflect EU average background conditions in terms of e.g. energy generation, material supply and transport. For both polymers, the main conversion process involved in the supply chain is steam cracking of naphtha and natural gas, delivering the monomers Ethylene, Propylene, Butadiene and other relevant intermediate flows (e.g. pyrolysis gas including mixtures of Benzene, Toluene and Xylenes). The inventory of this process is mainly based on industry data, completed, where necessary, by literature data. 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 co-products. The final polymerisation process of PP and LDPE is mainly modelled based on relevant industry data from internationally adopted production processes, integrated with literature data when needed. No allocation is performed for this stage, being PP and LDPE the only outputs of the respective production process.

#### **5.4.2.2 Bio-based polymers**

##### **5.4.2.2.1 Bio-based LDPE**

The production of bio-based LDPE out of Brazilian sugarcane was modelled based on an aggregated dataset provided by Thinkstep. The dataset covers the steps of sugarcane processing to bioethanol in Brazil (fermentation and distillation), bioethanol transport to Europe via transoceanic ship, conversion to bio-Ethylene (via dehydration), and its subsequent polymerisation to LDPE. 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. For the remaining conversion and polymerisation processes, no allocation nor substitution needs to be applied. The inventory is mainly 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.

##### **5.4.2.2.2 Polylactic Acid (PLA)**

For PLA production from US Maize, an aggregated, gate-to-gate inventory provided by Thinkstep was applied. The inventory includes the processes of maize wet milling for starch production, starch hydrolysis to glucose, its fermentation to lactic acid, oligomerization of lactic acid to lactide monomer, and final polymerisation of the latter to PLA. For the maize wet milling process, economic allocation is applied to the different co-products, while for downstream conversion processes no allocation is required. The overall inventory is mainly based on industry data from internationally adopted production processes, and is completed, where necessary, by literature data. It is noted

that the dataset reflects European background conditions for energy and material inputs, despite the analysed packaging film scenario assumes PLA production taking place in the US and subsequent transport of polymer resin to Europe. This is acknowledged as a limitation of this study.

#### 5.4.2.2.3 Starch (TPS)/PBAT blend

Thermoplastic starch used as a copolymer in starch-based food packaging 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 methacrylate<sup>54</sup>. 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. 8.33% each).

Starch production via wet milling of the different starch crops (maize, wheat and potatoes) was modelled based on life cycle inventory data reported for the corresponding processes in the Agri-footprint database (v 4.0), combined with background EF-compliant datasets for individual inputs and outputs. In these inventories, the allocation of process burdens to the different co-products (e.g. from corn wet milling) is based on the respective economic value, consistently with the fully vertically aggregated EF datasets available for starch production via wet milling (which are developed by the same data provider). Maize starch-related 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 approximated with data related to the production of Glycerine, derived from an aggregated, cradle-to-gate EF dataset. Conversely, the inventory of Glucose production was based on the *ecoinvent* dataset "[RER] Glucose production", which refers to pure Glucose produced via enzymatic hydrolysis of maize starch. However, in the implementation in the model, the maize starch input was replaced with the EU-average mix of starches from European starch crops (consistently with starch used for the main polymer). 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 (for steam). For Glycidyl Methacrylate, an average inventory related to the production of an unspecified organic chemical was applied (i.e. the *ecoinvent* dataset [GLO] market for chemical, organic)<sup>55</sup>, in the absence of more specific data on the relevant substance.

Compounding of starch and additives to form TPS was assumed to take place in the same facility where the final starch-based polymer is produced by blending TPS and PBAT granulate. This preliminary compounding activity was not modelled separately, since the respective burdens are taken into account 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 synthesis (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* database. Inputs of PET precursors (Ethylene Glycol and PTA) were replaced with the three PBAT co-monomers, considering the specific consumption reported in Schrijvers et al. (2014). These are based on a 90% synthesis efficiency, and are equal to 0.41 kg for

<sup>54</sup> Based on information provided in the documentation of the GaBi dataset related to Thermoplastic Starch production.

<sup>55</sup> 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.

Butanediol, 0.37 kg for Adipic Acid, and 0.33 kg for PTA (all expressed per kg of PBAT). Cradle-to-gate production inventories for these co-monomers were derived from aggregated GaBi datasets, which in the case of Butanediol and Adipic Acid refer to German background conditions, in the absence of representative data for EU conditions. Background energy inputs were combined with relevant EF datasets, while for material inputs and outputs the original *ecoinvent* datasets were kept. Infrastructure processes related to chemical factory construction were removed, to improve reliability of LCIA results in the Ozone Depletion impact category. Transport of PBAT co-monomers to the polymerisation plant was also included in the dataset, considering the default transport scenario defined in the PEF context (and in the Method) for transferring of goods from suppliers to factories/users located in the EU. 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.

The inventory of the final compounding step of TPS and PBAT (including blending of raw materials, cutting, drying and cooling) was based on the *ecoinvent* dataset “[RER] Polyester-complexed starch biopolymer production”, which relies on calculations and extrapolations from highly aggregated background data from the environmental product declaration of a starch-based polymer (MaterBi). Compared to the original dataset, inputs of naphtha and natural gas were removed, as referring to the supply of raw materials and of process energy associated with the production of the unspecified fossil-based copolyester used in the original blend, which is here represented by PBAT and was modelled separately (as described above). Similarly, the default input of maize starch was replaced with the average mix of starches from relevant European crops considered in this study (Section 5.1). Input quantities of starch, PBAT, and starch additives were set or adjusted so as to reflect the respective share in the final polymer blend (i.e. 30% starch, 10% starch 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 and waste flows reported in the original inventory were combined with EF background datasets for the modelling of the respective burdens. Finally, infrastructure processes related to chemical factory construction were removed, to improve reliability of LCIA results in the Ozone Depletion impact category.

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

#### 5.4.2.2.4 PLA/PBAT blend

PLA production from US maize was modelled based on an aggregated, gate-to-gate inventory dataset provided by Thinkstep, as already described in Section 5.4.2.2.2. The dataset reflects European background conditions for energy and material inputs, despite the analysed packaging film scenario assumes PLA production taking place in the US, and subsequent transport of polymer resin to Europe for further compounding with PBAT to form the final copolymer. This is acknowledged as a limitation of this study.

Modelling of PLA transport to Europe for compounding was made according to the default scenario specified in the PEF context (and in the present method) for import of goods from suppliers located outside Europe. A transoceanic ship transport was thus assumed as main route, considering a case-specific distance of 6,000 km (for the route New York – Rotterdam)<sup>56</sup>. In addition, a default distance of 1000 km was assumed to be covered by lorry (total weight >32 t; Euro 4) for road transport of the polymer resin to and from the harbour in the US or EU.

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<sup>56</sup> Defined based on the calculation tool provided by SeaRates.com, and available at <https://www.searates.com/services/distances-time>

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 route (i.e. esterification; Schrijvers et al., 2014). Polymerisation data were derived from the most recent implementation of the PlasticsEurope ecoprofile in the *ecoinvent* database (CPME, 2017), and were combined with GaBi or EF datasets for the production of the three PBAT co-monomers (1,4-Butanediol, Adipic Acid and Terephthalic Acid), as better detailed in Section 5.4.2.2.3. PBAT granules were then assumed to be transported to the compounding facility based on the default scenario specified in the PEF context (and in the Method) for transport of goods from suppliers to factories/users 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 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 polymeric blends, as reported in the *ecoinvent* dataset "[RER] Polyester-complexed starch biopolymer production". Only flows related to energy inputs, waste generation and direct emissions were maintained from the original inventory, and combined with EF background datasets for the modelling of the respective burdens. Inputs related to raw material supply (i.e. starch, naphtha and natural gas) were instead replaced with relevant raw material inputs (i.e. PLA and PBAT), modelled as described above. A 100% conversion efficiency was assumed for the process, considering that material losses during compounding of starch-based polymer blends were reported to be negligible (Broeren et al., 2017). Infrastructure processes related to chemical factory construction were removed, to improve reliability of LCIA results in the Ozone Depletion impact category.

#### 5.4.2.3 CO<sub>2</sub>-based PP

Conversion of captured CO<sub>2</sub> to Methanol via hydrogenation was modelled based on raw material (CO<sub>2</sub> and H<sub>2</sub>) and energy consumption data reported by Hoppe et al. (2017), which were in turn derived from the results of the process simulation exercise performed by Rihko-Struckmann et al. (2010). Similarly, the inventory of Propylene production through the Methanol-to-olefin route was developed based on data derived from the process simulation results by Xiang et al. (2014), again as reported in Hoppe et al. (2017). In both inventories (summarised in Table 5.4) energy inputs were combined with EF background datasets for the modelling of the respective burdens. According to the approach applied as a base case to the modelling of the CO<sub>2</sub> utilisation pathway (i.e. the Circular Footprint Formula with a value of the A factor equal to 0.5; Section 5.4.1.3), only 50% of the burdens of the Methanol and Propylene synthesis processes were assigned to the CO<sub>2</sub>-based product (i.e. Propylene). However, the latter is assigned 50% of the primary production burdens of replaced fossil-based Propylene via naphtha cracking. This process was modelled based on a modified version of the aggregated, cradle-to-gate GaBi dataset "[EU-28] Propene (propylene) steam cracker | single route, at plant | 1.91 kg/m<sup>3</sup>, 42.08 g/mol", which consistently relies on the same methodological choices and data sources as the gate-to-gate dataset applied to the modelling of PP production, and already described in Section 5.4.2.1. However, upstream burdens associated with the supply of naphtha and of the related fossil-based feedstock were purposefully subtracted and modelled separately as described in Section 5.4.1.1.

The final polymerisation step to Polypropylene was modelled based on data from the latest PlasticsEurope ecoprofile (PlasticsEurope, 2016b), as implemented in the *ecoinvent* database. Inventory data limited to the polymerisation stage could not be extracted from the partially aggregated Thinkstep dataset applied to model fossil-based PP production. Therefore, it was not possible to perform a consistent modelling of this process across all the investigated PP-based scenarios (i.e. fossil-based and CO<sub>2</sub>-based PP). The applied data represent average values of data collected from several production units in Europe operated by PlasticsEurope member companies, covering 76% of the total production capacity in the same region. The mix of commercial PP production technologies is considered, including slurry suspension polymerisation, bulk suspension polymerisation,

and gas phase polymerisation (using Ziegler-Natta and Metallocene catalysts). The final 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 production<sup>57</sup>. No allocation or substitution was performed, since the process is mono-functional, delivering only PP as a product.

All process steps from Methanol synthesis to polymerisation were assumed to take place in the same production facility (or in proximate ones). Therefore, no specific transport activities were modelled across these stages.

**Table 5.4.** Life cycle inventories of the processes involved in the conversion of captured CO<sub>2</sub> and Hydrogen into CO<sub>2</sub>-based Propylene (Methanol synthesis and conversion to Propylene via the Methanol-to-olefin route).

Flow	Amount	Unit	Dataset	Database
<b><i>Methanol synthesis (CO<sub>2</sub> hydrogenation) – per 1 kg of Methanol</i></b>				
Captured CO <sub>2</sub>	1.374	kg	-	-
Hydrogen	0.189	kg	-	-
Electricity	1.271	kWh	[EU-28+3] Electricity grid mix 1kV-60kV; AC, technology mix   consumption mix, at consumer   1kV - 60kV {34960d4d-af62-43a0-aa76-adc5fc57246}	EF
Heat (output) <sup>(1)</sup>	0.36	MJ		
<b><i>Propylene synthesis (Methanol-to-olefin) – per 1 kg of Propylene</i></b>				
Methanol	2.571	kg	-	-
Electricity	0.458	kWh	[EU-28+3] Electricity grid mix 1kV-60kV; AC, technology mix   consumption mix, at consumer   1kV - 60kV {34960d4d-af62-43a0-aa76-adc5fc57246}	EF
Heat (steam) <sup>(2)</sup>	5.587	MJ	[EU-28] Thermal energy, production mix	Lit. + EF

<sup>(1)</sup> Output heat from the chemical exothermic reaction. Not modelled in the LCI, due to the relatively low amount.

<sup>(2)</sup> Dataset developed based on IEA statistics for heat generation by source in the EU (IEA, 2019), combined with EF background datasets for heat production from each source included in the mix (i.e. 42.4% natural gas, 30.8% hard coal, 21.8% biomass, 5% heavy fuel oil).

#### 5.4.2.4 Transport of polymer granulate to the article production site

Modelling of transport of polymer granulate from the polymerisation or compounding plant (inside or outside the EU), to the food packaging film manufacturing site in Europe, is based on the default transport scenario (distances and vehicle types) specified in the PEF context (and in this method) for the route supplier-to-factory. In the case of polymers produced in Europe (i.e. all the polymers and copolymers investigated in this case study, except for PLA and the imported share of fossil-based PP and LDPE), the following routes were thus considered:

- (i) 130 km by lorry (total weight >32 t; Euro 4);
- (ii) 240 km by train (average freight); and
- (iii) 270 km by ship (barge).

<sup>57</sup> To overcome issues of reliability of LCIA results in the Ozone depletion impact category, infrastructure processes related to the polymerisation plant ([RER] *Chemical factory construction, organics*) were removed. For the same reason, a number of chemicals were removed from the proxy dataset used to model the production of unspecified organic chemicals ([GLO] *Market for chemical, organic*), i.e. Acetic Acid, Methanol, Urea, Vinyl Acetate, Ethylene Dichloride, and Formaldehyde). The respective shares were then equally subdivided among the remaining chemicals.

For the imported share of fossil-based PP and LDPE (10% and 14%, respectively), a transoceanic ship transport was considered as the main transport route. The corresponding overall 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)<sup>58</sup>. Countries contributing to at least 90% of the overall import were considered in the calculation, for an overall distance equal to 8,609 km for fossil-based PP, and to 6,528 km for fossil-based LDPE (see Tables 5.5 and 5.6). Oceanic ship transport was complemented with road transport to and from the harbour in both the exporting and importing country (i.e. the EU). Road transport is made by lorry (total weight >32 t; Euro 4) along an overall default distance of 1000 km.

A similar transport scenario to that of imported fossil-based polymers was considered also for PLA. However, in this case, an overall sea distance equal to 6,000 km was assumed, based on the estimated mileage from the SeaRates calculation tool mentioned above for the suggested route New York – Rotterdam. A 1000 km road transport by lorry (>32 t; Euro 4) was again assumed for the route to and from the harbour in the exporting country (US) and in Europe.

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

**Table 5.5.** Calculation of the overall average sea distance for imports of fossil-based PP to Europe.

Exporting country	Import (¹) (%)	Import (% cum.)	Distance (²) (km)	Weighted distance (km)
SAUDI ARABIA	41.3	41.3	8767.28	3622
KOREA, REPUBLIC OF (SOUTH KOREA)	13.5	54.8	16702.31	2259
INDIA	7.15	62.0	10267.28	734
RUSSIAN FEDERATION (RUSSIA)	6.75	68.7	5574.44	376
BRAZIL	6.63	75.4	10107.97	670
ISRAEL (GAZA and JERICHO->1994)	4.86	80.2	2987.48	145
SOUTH AFRICA (incl. NA ->1989)	4.26	84.5	12160.92	518
EGYPT	3.20	87.7	3212.14	103
UNITED STATES	2.99	90.7	6061.9	181
Other countries	9.32	100	-	-
<i>Overall weighted distance</i>				<i>8,609</i>

(¹) Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

(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 Middle-East and Asia were determined considering Marseille as destination port in Europe. For imports from other countries (in the case of PP Russia, Brazil, South Africa and United States), Rotterdam was considered as destination port.

<sup>58</sup> Available at: <https://www.searates.com/services/distances-time/>

**Table 5.6.** Calculation of the overall average sea distance for imports of fossil-based LDPE to Europe.

Exporting country	Import <sup>(1)</sup> (%)	Import (% cum.)	Distance <sup>(2)</sup> (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	-	-
Overall weighted distance				6,528

<sup>(1)</sup> Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

<sup>(2)</sup> 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 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.

### 5.4.3 Article Production Stage

For all the investigated materials, food packaging film was assumed to be manufactured via blown film extrusion. This is the most common method for large scale production of plastic films, and the one facilitating most the introduction of biaxial orientation into the film (Crawford and Martin, 2020), such as in biaxially oriented PP and PLA. In blown film extrusion, melted plastic granules are passed through an annular die to form a thin tube, which is inflated with air from the 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 bag production (Crawford and Martin, 2020).

The inventory of the film extrusion process was derived from the aggregated, EF dataset “[EU-28+EFTA] Film Extrusion (blowing); plastic extrusion | production mix, at plant | for PP, PE, PVC, PET and PS”, which accounts for a 99% conversion efficiency. Process losses (e.g. flawed film) 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 PP granules were assumed to replace virgin, fossil-based PP granules, since no bio-based PP was reported to be available on the market at the time of developing this analysis (EUBP, 2019). Similarly, recycled LDPE granules (both fossil-



based and bio-based) replaced virgin, fossil-based LDPE granules, being the estimated share of bio-based LDPE on the marked very low (i.e. 0.2%). 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 PP and LDPE film, in the absence of more specific data for recycling of pre-consumer, industrial scraps. For further detail on the modelling of recycling and avoided virgin material production (including the implementation of the Circular Footprint Formula), the reader is hence referred to Section 5.4.5 on End of Life modelling.

Transport of packaging film from the manufacturing site to food production/processing and packing facilities was also included in the Article Production stage, and modelled according to the default transport scenario specified in the PEF context (and in the present 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.

#### 5.4.4 Distribution Stage

The transport of food packaging film (converted into a final packaging item) from the food packing site to the final user was modelled based on the default transport scenario specified in the PEF context and in this method for the route *factory* → *retail* → *final client*. The following routes were thus considered:

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

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

#### 5.4.5 End of Life Stage

This section describes the modelling of the End of Life stage of food packaging film. In particular, Section 5.4.5.1 focuses on the definition of the EU-average End of Life scenario, which is considered as a base case for the calculation of the potential impacts of the compared LCA scenarios. The remaining sections (5.4.5.2 – 5.4.5.7) address the modelling of waste collection and transport, and of the different End of Life options explored in the study. Finally, Section 5.4.5.8 provides case-specific details on the estimate of the potential contribution of food packaging film to macro-plastics formation at the End of Life (including product litter) and to micro-plastics generation throughout the supply chain.

##### 5.4.5.1 End of Life scenario

The same EU-average End of Life scenario was considered for PP-based and LDPE-based food packaging films, regardless of the type of feedstock used for polymer production (fossil sources, biomass or captured CO<sub>2</sub>). Indeed, the type of feedstock does not affect the viable End of Life options for packaging film made of these materials, nor the options currently applied to it, which include mechanical recycling, incineration and landfilling.

These options were applied at the following rates in the modelled End of Life scenario: 20% mechanical recycling, 42% incineration, and 38% landfilling. The share of post-consumer packaging film separately collected for recycling was defined according to the EU-average recycling rate estimated for flexible LDPE/LLDPE products by Plastics Recyclers Europe (i.e. 20%; PRE, 2019). The same rate was also applied to PP-based films, in the absence of specific data for films made of this material. While it is more likely that recycling successfully takes place for (non-contaminated) packaging film used

in non-food applications (e.g. stretch film or heat-shrink film used as transport or secondary packaging), the estimated recycling rate was assumed to uniformly apply to all types of packaging films (including those used in food applications), since no distinction among different applications is made in the abovementioned source. This potentially optimistic assumption is anyway considered to only marginally affect the LCA results, being the applied recycling rate quite limited (20%). Note, also, that being this rate specifically representative of flexible (packaging) film, its application was preferred over the default recycling rate (R2 value) reported in Annex C of the PEF method for generic plastic packaging (and to the material-specific rates reported for PP and PE in the same Annex).

For incineration and landfilling, EU-average rates for total plastic packaging waste were estimated and applied, in the absence of specific data for flexible packaging film. The estimate was based on statistics related to plastic packaging waste management in Europe for the years 2014-2016 (Eurostat, 2019b). Assuming that all the packaging waste sent to recovery operations other than recycling is incinerated, it was possible to estimate an average incineration rate equal to 31%. Similarly, assuming that all the waste that is not recovered is landfilled, an average landfilling rate of 28% could be estimated. In relative terms, 53% of total plastic packaging waste that is not recycled is incinerated, while 47% is landfilled. The share of non-recycled LDPE or PP film (80%) was thus assumed to be routed to incineration and landfilling according to these proportions (i.e. 42% incineration and 38% landfilling).

For biodegradable packaging films (PLA, Starch-based and PLA-based), currently viable End of Life options include, beyond incineration and landfilling, also biological treatment alternatives, i.e. composting and anaerobic digestion<sup>59</sup>. Mechanical recycling was instead not considered a viable option for packaging film made of these materials, as they are primarily intended as biodegradable alternatives allowing separate collection of (food-contaminated) packaging items with organic waste, potentially facilitating their management at End of Life. Moreover, even if they were separately collected with conventional plastic waste, they would not be sorted out for recycling with the current sorting equipment, and flotation in hot caustic baths (normally used for plastic separation and cleaning during recycling) would likely initiate or further promote material degradation, making them unavailable for actual recycling.

For these biodegradable alternatives, the EU-average End of Life scenario was defined assuming that the film is separately collected with organic waste for biological treatment with the same efficiency as conventional (non-biodegradable) film separately collected for recycling (i.e. 20%, as discussed above). This share was then split between composting (18%) and anaerobic digestion (2%), considering that over 90% of separately collected organic waste in the EU is processed into compost (ECN, 2019), the rest being handled through anaerobic digestion processes. Finally, non-separately collected packaging film was assumed to be routed to incineration and landfilling according to the same proportions estimated above for conventional films, i.e. 42% incineration and 38% landfilling.

#### **5.4.5.2 Modelling of waste collection and transport**

Collection and transport of post-consumer food packaging film for mechanical recycling, incineration or landfilling was modelled according to the pathways, vehicle types and distances reported in Rigamonti et al. (2013) for either separately collected plastic waste, or residual waste, at the municipal level. These data refer to a region with a well-

<sup>59</sup> Degradation of biodegradable polymers under anaerobic digestion conditions is typically low, and a further aerobic treatment step (post-composting) is normally needed to achieve suitable degradation levels. In addition, current composting standards for plastic materials and products (e.g. EN 13432 and EN 14995) provide only partial requirements for anaerobic digestion, i.e. in terms of minimum mineralisation level to be achieved during the process. Conversely, other relevant aspects (such as disintegration requirements and quality of the output material) are not addressed for anaerobic biodegradation. However, anaerobic digestion was included in the EU-average End of Life scenario since the organic waste stream where biodegradable plastic articles are collected follows also this route at the EU level.

1 developed waste management scheme in northern Italy, and can be considered  
2 representative of several regions in Europe where good levels of separate collection are  
3 achieved, with the implementation of kerbside collection systems. While this may not still  
4 be a common practice across all Europe, the approximation is considered reasonable, due  
5 to the typically moderate contribution of waste collection and transport to the overall End  
6 of Life (and lifecycle) impacts (Rigamonti et al., 2014). More information on the main  
7 modelling assumptions and details is provided in Section 4.4.5.2. Note that when EF  
8 incineration and landfilling datasets were used, no transport of collected waste was  
9 separately modelled, since the respective burdens are already accounted for in such  
10 datasets. In this case, only collection was modelled, according to the approach described  
11 above.

12 Relevant data and assumptions for the modelling of separate collection of biodegradable  
13 packaging films (as part of the organic waste stream), and of their subsequent transport  
14 to composting or anaerobic digestion, were derived as well from Rigamonti et al. (2013).  
15 These were implemented in the model as detailed in Table 5.7, relying on EF-compliant  
16 datasets for background inventories related to the use of vehicles for collection and  
17 transport.

18

**Table 5.7.** Modelling of collection of source-separated biodegradable plastic articles as part of the organic waste stream and of their subsequent transport to biological treatment <sup>(1, 2)</sup>.

Collection type	Share (%)	Distance (km/t)	Vehicle	Share (%)	Dataset	Amount (km*t/t <sub>collected waste</sub> )
Kerbside	100%	37.6	Medium/large-sized truck	13%	[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	5.0
			Small-sized truck	87%	[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	32.6
Transport to composting and anaerobic digestion						
Transport to composting	100%	54.5	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	54.5
Transport to anaerobic digestion	100%	64.7	Large truck	100%		64.7

<sup>(1)</sup> Composting and anaerobic digestion.  
<sup>(2)</sup> Based on Rigamonti et al. (2013).

### 5.4.5.3 Modelling of sorting and recycling

Before recycling, bales of separately collected plastic waste containing post-consumer food packaging film are firstly 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.

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 data collected from different dual-stream and single-stream sorting facilities in the Country). For implementation in the model, the reported input and output data were complemented with background EF datasets representative of EU-average conditions, as better detailed in Section 4.4.5.3 (Table 5.8). A 100% sorting efficiency was assumed for waste packaging film, which entirely consists of one of the targeted materials for recycling. Moreover, no burdens from the treatment of any impurities sorted out as rejects were assigned to sorting of waste packaging film, to avoid falsely “punishing” it with burdens from unrelated product waste (and mostly coming from incorrect citizen behaviour). Therefore, the modelled sorting process only accounted for the material and energy inputs required to carry out this waste treatment activity.

For mechanical recycling of PP-based food packaging films, a new dataset was developed based on foreground inventory data available as well in Franklin Associates (2018), combined with EF background datasets for energy and material supply under EU-average conditions<sup>60</sup>. The inventory is based on mass-weighted average values of data collected from several recycling facilities in the United States. While these data are likely more representative of recycling of PP used in rigid or semi-rigid applications (e.g. trays or bottles), they were considered suitable to approximate recycling of flexible PP, which is expected to rely on similar unit operations (e.g. grinding/shredding, washing/flotation and granulation). The inventory accounts for an overall recycling efficiency equal to 85.5%, with removed contaminants being sent to incineration along with process waste. This is in line with the typical fate of recycling residues, which due to their high calorific value are normally sent to incineration or co-combustion in cement kilns (Rigamonti et al., 2014).

As for LDPE-based packaging film recycling, no specific inventory data are available. Therefore, this process was modelled through an aggregated, EF-compliant 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)<sup>61</sup>. The dataset is developed based on literature data for these unit operations, and accounts for an overall recycling efficiency equal to 84% (on the input material). Process waste and scrap are sent to incineration, consistently with the typical fate of plastic recycling residues, and the assumption performed in the PP recycling process. Similarly to the latter, the applied dataset is likely more representative of recycling of rigid or semi-rigid plastic products, such as bottles and trays. However, it was considered a suitable approximation also for flexible products (i.e. packaging film in this case), since their recycling is expected to rely on similar unit operations to those considered in the applied dataset.

Recycled polymer granulate was assumed to replace virgin granulate of the same material, whose primary production burdens were credited to the system. For polymers

<sup>60</sup> An aggregated EF-compliant dataset is also available for PP recycling in the US (based on an older version of the same inventory data source, i.e. Franklin Associates, 2011). However, since this dataset could not be adjusted to reflect EU background conditions (due to its aggregated nature) and most recent data are available, a new disaggregated dataset was developed.

<sup>61</sup> [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.

having both a fossil-based and a bio-based alternative available on the market as of today (i.e. only LDPE) 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 for LDPE, being equal to only 0.2% (while 99.8% of LDPE available on the market is fossil-based)<sup>62</sup>. To account for the lower overall quality of recycled polymers compared to the replaced virgin polymers, a substitution ratio equal to 0.75 was considered for LDPE, according to default values specified in the PEF context for LDPE derived from (packaging) film recycling. The same ratio was also applied, for consistency, also to recycled PP granules replacing virgin PP granules, in the absence of specific values for PP derived from film recycling, and to prevent distorted comparison between LDPE and PP packaging film. To model the burdens of avoided primary production, the same datasets (or combination of datasets) used for the modelling of upstream production of the relevant polymer were applied (as described in Sections 5.4.2.1, and 5.4.1.1 for the related feedstock). This was made for consistency reasons, and to avoid possible distortions by applying different datasets from other sources.

According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the sorting and recycling processes were allocated to the system ( $A = 0.5$  for PP and PE used in unspecified applications and for generic plastic packaging). Similarly, only 50% of the benefits from avoided virgin material production were assigned to the system itself.

#### **5.4.5.4 Modelling of incineration**

For conventional fossil-based polymers (i.e. PP and LDPE) aggregated, material-specific incineration datasets are available from the EF database, and were applied to model the treatment of food packaging film made of these materials in a municipal waste incineration plant. Similarly, for bio-based LDPE and neat PLA film, as well as for TPS and PLA used as copolymers in starch-based and PLA-based packaging film, partially aggregated, material-specific inventories from the GaBi database were applied (no EF datasets are available for these polymers).

All the selected datasets are developed based on a waste-specific incineration model, which has been described more in detail in Section 4.4.5.4. 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 general approach to handle energy recovery situations adopted in the PEF context and in the Method (Report I), the product system generating the waste material sent to incineration (i.e. the packaging film life cycle, in this case) takes the full burdens from the incineration process. However, it is credited for 100% of the benefits from avoided production of conventional energy (electricity and heat) replaced by energy recovered from waste. While in EF incineration datasets these credits are already accounted for in the aggregated inventory, for GaBi datasets they were added to the main process inventory. Therefore, the EU residual electricity grid mix (as modelled in the EF 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 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 includes 42.4% natural gas, 30.8% hard coal, 21.8% biomass, and 5% heavy fuel oil. In the calculation of these

<sup>62</sup> The share of bio-based LDPE consumed in Europe was estimated based on the global production capacity of bio-based PE in 2018 (European Bioplastic, 2019; 200 kt) and of PE as a whole in 2016 (PlasticsInsight, 2019; 103 Mt).

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. Also, thermal energy from waste (11%) was excluded, as according to the approach described above for the handling of energy recovery situations, 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 PLA-based and starch-based packaging film, no ready-to-use incineration datasets were available. A disaggregated, material-specific inventory was thus developed, based on the most recent version of the calculation tool developed by Doka (2009a) for the modelling of material incineration within municipal solid waste incineration plants. The model operates similarly to the one used for the development of the EF 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 5.8 in Section 5.4.5.5 for an overview of the considered PBAT 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<sub>x</sub> control technologies, and a few other specific parameters. In this case 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, and applied in the modelling. These efficiencies account for the share of waste routed to incineration plants operating without any energy recovery (estimated to be 9% for municipal waste)<sup>63</sup>, while considering that plants with energy recovery operates with an average gross electricity efficiency equal to 15.1%, and a gross thermal efficiency equal to 35% (CEWEP, 2012)<sup>64</sup>. Credits associated with recovered energy were calculated as described above for existing GaBi datasets (i.e. electricity from the EU-average 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, background EF datasets were applied.

#### **5.4.5.5 Modelling of landfilling**

Landfilling of packaging film made of conventional, non-biodegradable polymers (i.e. fossil-based PP and LDPE) was modelled based on a common, aggregated EF dataset representing disposal of plastic waste in a managed, municipal solid waste landfill (*[EU-28+EFTA] Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific sites)*). The underlying inventory is material-specific, but refers to the average chemical composition and degradability values of generic plastic waste, rather than to those of the specific polymer. 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<sup>65</sup>. 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 emissions to the environment over a 100 year time horizon. Emissions occurring beyond 100 years are not accounted

<sup>63</sup> Calculated as the average share of municipal waste incinerated without energy recovery over the years 2015-2017 (Eurostat, 2019d).

<sup>64</sup> 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).

<sup>65</sup> While more polymer-specific landfilling datasets are available from other databases (i.e. *ecoinvent*), the mentioned EF dataset for landfilling of generic plastic was selected, as specifically referring to EU as the reference geography (in contrast to available polymer-specific datasets), and to keep consistency with the overall dataset selection "hierarchy" followed in this project (Section 3.5).

for in the model. Landfill gas generation is calculated based on the organic carbon content in the waste and the respective degradation rate assumed over 100 years (not reported), while, 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 rate of landfill gas, energy efficiencies of engines, collection rate of leachate and respective treatment efficiencies). Further details on how these parameters were set to reflect the EU-average situation are provided in Section 4.4.5.5.

The same dataset described above for landfilling of generic plastic waste was also applied as a proxy for landfilling of packaging film made of “drop-in”, non-biodegradable, bio-based polymers, i.e. bio-based LDPE. Compared to the original dataset, emissions of CO<sub>2</sub> and CH<sub>4</sub> to air have been converted from fossil to biogenic emissions, to reflect the bio-based origin of carbon in such polymer. 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 amounts of fossil CO<sub>2</sub> and CH<sub>4</sub> reported as released to air were converted to biogenic emissions. This approximation is considered acceptable, as reported CO<sub>2</sub> and CH<sub>4</sub> emissions are modest, and overall amounting to only 1.4% of the carbon content in the landfilled plastic material, despite they also include the contribution of emissions from background activities. This can be partly explained by the generally low degradation (and mineralisation) of conventional, non-biodegradable polymers in landfill, which is typically assumed to be in the range of 1% over 100 years (Doka, 2009b). On the other hand, it is acknowledged that this approach is in favour of the bio-based polymer (i.e. Bio-LDPE).

In line with the time horizon applied for landfill emission modelling in the selected dataset, (biogenic) carbon in the landfilled polymer that is not degraded (mineralised) after 100 years from deposition was considered to be never released from the landfill body. As a base case, this missed release of atmospheric carbon taken up during biomass growth is not reflected in Climate Change LCIA results for bio-based LDPE food packing film, since characterisation factors for biogenic CO<sub>2</sub> flows have been set to zero (fully conforming to provisions in the PEF method). However, to show the relevance of this choice on the overall results, the Climate Change impact of the mentioned bio-based alternative was also calculated by acknowledging the effects of non-released biogenic carbon. For this purpose, a specific CO<sub>2</sub> uptake was modelled in the inventory, expressing the net amount of CO<sub>2</sub> taken up during biomass growth and not released (as CO<sub>2</sub> or CH<sub>4</sub>) during polymer degradation in landfill. The uptake was calculated based on the biogenic carbon content in the polymer, and assuming a mineralisation rate over 100 years equal to 1% (Doka, 2009b). While the mineralisation rate assumed in the dataset is not reported explicitly, it seems to be in line with this assumption (according to the figures reported above regarding the inventoried carbon emissions). Considering a biogenic carbon content in Bio-LDPE equal to 81.9%, a carbon uptake equal to 2.97 kg CO<sub>2</sub>/kg Bio-LDPE was thus calculated. This uptake was then characterised with a factor of -1 kg CO<sub>2</sub> eq. per kg CO<sub>2</sub>-C not degraded.

For packaging film made of non-“drop-in” (partially) bio-based and biodegradable polymers (i.e. PLA, starch-based, and PLA-based), a material-specific landfilling inventory was built, based on the calculation tool developed by Doka (2009b) for the modelling of waste disposal into sanitary landfills<sup>66</sup>. Similarly to waste incineration, the tool allows to calculate material-specific landfilling inventories accounting for the specific chemical composition and other relevant chemo-physical properties of the landfilled waste (Table 5.8), as well to adjust a number of relevant site- and technology-specific parameters to the reference geography and to the corresponding average landfilling technology. For this purpose, the model applies element-specific transfer coefficients to calculate the distribution of decomposition products originating from elements in the waste

<sup>66</sup> Note that for starch-based and PLA-based packaging film, separate inventories were developed for the single copolymers (i.e. TPS, PLA and PBAT).



composition between landfill gas and leachate, and their ultimate emissions 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.

**Table 5.8.** Elemental composition and lower heating value of PLA, TPS and PBAT considered for End of Life modelling (based on literature or stoichiometry).

Element	Share (%)	Element	Share (%)
<b>PLA <sup>(1)</sup></b>			
TS	100	C biogenic (%TS)	49.5
Water	0	H (%TS)	5.60
VS (%TS)	100	O (%TS)	44.5
Ash (%TS)	0	N (%TS)	0.1
C fossil (%TS)	-	S (%TS)	0.3
LHV = 18.4 MJ/kg <sup>(2)</sup>			
<b>TPS <sup>(3)</sup></b>			
TS	100	C fossil (%TS)	-
Water	0	C biogenic (%TS)	44.8
VS (%TS)	100	H (%TS)	6.58
Ash (%TS)	0	O (%TS)	48.6
LHV = 17.3 MJ/kg <sup>(2)</sup>			
<b>PBAT <sup>(4)</sup></b>			
TS	100	C fossil (%TS)	53.7
Water	0	C biogenic (%TS)	-
VS (%TS)	100	H (%TS)	6.47
Ash (%TS)	0	O (%TS)	39.8
LHV = 21.1 MJ/kg <sup>(2)</sup>			

<sup>(1)</sup> Based on results from composition analysis related to Ingeo polymer available at: <https://www.natureworksllc.com/What-is-Ingeo/Where-it-Goes/Incineration>

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

<sup>(3)</sup> Based on stoichiometry, and considering that TPS consists of 75% starch and 25% plasticisers/additives (equally split among Glycerol, Sorbitol and Glycidyl Methacrylate).

<sup>(4)</sup> Based on stoichiometry.

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. carbon to CH<sub>4</sub> and CO<sub>2</sub>) within the landfill. Degradability values applied in the modelling for each material are summarised in Table

5.9. For neat PLA film, a 1% biodegradation was assumed, according to the results from Accelerated Landfill Conditions (ALC) tests simulating approximately 100 years of deposition in a biologically active landfill (390 days at 21 °C, Kolstad et al., 2012). These results showed no significant biodegradation of both amorphous and semi-crystalline PLA over such a time horizon, even at the highest humidity level applied for testing (i.e. 65%, which should be considered in a conservative approach). Therefore, the same value of degradability assumed in the Doka (2009b) model for conventional, non-biodegradable polymers such as PET and PE was consistently applied (i.e. 1%). Similarly, for starch-based film (or better, its copolymers TPS and PBAT), results from ALC testing of Mater-Bi (NF 803 grade) at 65% humidity were considered (Vermeulen, 2007). In this case, a biodegradation of nearly 11% was observed at the end of the test (based on net biogas production), although the testing timeframe was lower than above (i.e. 180 vs 390 days) and the degradation curve (evolution of net biogas production) had not yet reached a plateau, but was still increasing. Hence, a higher biodegradation rate can actually be expected over a 100 year timeframe. Finally, in the absence of specific data related to the biodegradation of PLA/PBAT films under landfill conditions, a rate identical to that assumed for starch-based film was applied.

As discussed above for Bio-LDPE film, (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, to calculate also in these cases a variant of Climate Change LCIA results that acknowledge the effects of non-released atmospheric carbon, a specific CO<sub>2</sub> uptake was modelled in the inventory, expressing the net amount of CO<sub>2</sub> taken up during biomass growth and not released (as CO<sub>2</sub> or CH<sub>4</sub>) during polymer degradation in landfill. Considering the biogenic carbon contents reported in Table 5.8, the following carbon uptakes were calculated, based on the mineralisation rates reported above: 1.80 kg CO<sub>2</sub>/kg neat PLA, 1.62 kg CO<sub>2</sub>/kg PLA blended with PBAT (i.e. 1.13 kg CO<sub>2</sub>/kg PLA/PBAT blend), and 1.46 kg CO<sub>2</sub>/kg TPS (i.e. 0.585 kg CO<sub>2</sub>/kg TPS/PBAT blend).

Site-specific and technology-specific parameters were set so as to reflect as much as possible the average situation at the EU level, as detailed in Section 4.4.5.5. These include the rates of landfill gas capture, utilisation and flaring, as well as the mean annual precipitation and temperature. For the other parameters (e.g. energy efficiencies of gas engines, mean annual evapotranspiration, landfill height and duration of the filling phase) the default values assumed in the model were kept.

In the final landfilling datasets, 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 database<sup>67</sup>. However, for energy-related flows, background EF datasets were applied.

**Table 5.9.** Degradability values considered for biodegradable food packaging films (PLA, starch-based, PLA-based) during the first 100 years from deposition in a managed landfill for municipal solid waste.

Polymer	Degradability (%)	Source
PLA (neat)	1	Kolstad et al. (2012)
TPS/PBAT blend <sup>(1)</sup>	11	Vermeulen (2007)
PLA/PBAT blend <sup>(1)</sup>	11	- <sup>(2)</sup>

<sup>(1)</sup> Landfilling of the two copolymers was modelled separately, by consistently applying to both of them the same biodegradability value of the overall polymer blend.

<sup>(2)</sup> The same biodegradability value as TPS/PBAT blend was applied, in the absence of specific data for PLA/PBAT blends.

<sup>67</sup> Exceptions are the inputs of diesel (burned in building machine) and pitch, which were replaced with suitable EF datasets, to improve reliability of LCIA results in the Ozone Depletion impact category.

#### **5.4.5.6 Modelling of composting and compost use-on-land**

A waste-specific life cycle inventory was developed for aerobic composting of biodegradable food packaging films (i.e. PLA, Starch-based and PLA-based films). The modelling was carried out according to the general recommendations specified in the Method (Report I - Sections 4.4.12.4 and 4.4.12.7), while relying on process-specific burdens and element-specific transfer coefficients available in the EASETECH model (Clavreul et al., 2014) for the chosen composting technology (i.e. tunnel composting, as discussed below). The elemental composition assumed for each packaging film material is the one already reported in Table 5.8, while the applied biodegradation rates are summarised in Table 5.10. The latter were defined as the average of (experimental) values available in the literature for films or similar plastic articles. Since these values were at least partly determined under real composting or laboratory conditions, they were preferred to the application of the minimum biodegradation rate required by the European standard for compostable plastic packaging (EN 13432) i.e. 90%. This has indeed to be proven under optimised laboratory testing conditions, and may not be achieved in real composting plants (or it may anyway differ from values obtained from real laboratory testing available in the literature).

An enclosed composting facility (tunnel composting) was considered for modelling purposes, since biodegradable plastic articles are typically collected along with food waste from households (rather than garden waste), which is preferably treated within enclosed facilities (Krogmann et al., 2011). The modelled process includes a first screening step, where items larger than 100 mm are normally removed. In this stage, 30% of waste packaging film entering the composting plant was assumed to be removed, regardless of its specific shape or size at that point in time, and sent to incineration (modelled as described in Section 5.4.5.4).

Carbon in biodegradable packaging films sent to composting was assumed to mineralise according to the biodegradation rates reported in Table 5.10. The same rates were also assumed for biodegradation of volatile solids (VS) in each film material. According to the emission factors (transfer coefficients) reported in the EASETECH model, 99.8% of mineralised carbon is converted to CO<sub>2</sub>, the rest (0.2%) being converted to CH<sub>4</sub>. The latter is then mostly oxidised to CO<sub>2</sub> (95%), while only 5% is eventually emitted as Methane, equalling 0.01% of the mineralised carbon.

No waste-specific Nitrogen emissions from the composting process were modelled, being the Nitrogen content of the investigated biodegradable polymers equal to zero or negligible (e.g. for PLA). Process-specific emissions (H<sub>2</sub>S and Terpenes) were however included in the inventory (i.e. 0.1929 mg and 1.222 mg per kg of input waste, respectively). Additional process specific burdens accounted for in the model include electricity supply for all the operations carried out at the plant (0.053 kWh/kg input waste), as well as diesel usage for material handling in the composting facility (0.001 litre/kg input waste).

The amount of residual composted material (biomass and simpler organic compounds) obtained from the biodegradation of the films was calculated as the sum of non-degraded volatile solids, as well as of ash and water<sup>68</sup> originally included in the material sent to composting. The residual material was assumed to be entirely applied on agricultural land, where biodegradation of the residual material further advance. Application is made by means of tractors, with a specific diesel consumption equal to 0.00042 l/kg composted material applied. The emissions from on-land application were modelled based on average emission factors estimated by Bruun et al. (2006), considering a time horizon of 100 years from application. These factors are derived from simulations of on-land application of municipal organic waste compost under Danish conditions (in terms of crop

<sup>68</sup> It is acknowledged that the water content in the residual composted material may change compared to the original material sent to composting due to losses and integration during the process. However, the water content of the investigated biodegradable polymers is equal to zero, so that this simplifying assumption does not affect the calculated amount of residual material from composting.

rotation, soil type and climate), carried out via an agro-ecosystem model. Considering average factors across all types of temperate soils, 89.3% of the applied carbon was assumed to mineralise to Carbon Dioxide over 100 years, 0.01% was released as Methane, while the remaining 10.7% was not released (being part of more stable organic compounds)<sup>69</sup>. Nitrogen and Phosphorus emissions were not calculated, being the content of these elements in the examined polymers and in the resulting composted material equal to zero or negligible. Similarly, no heavy metal emissions were modelled, as no metals are included in the material composition.

Since no nutrients are supplied to the soil with the residual composted material derived from film biodegradation, no mineral fertilisers were assumed to be replaced after its application. Hence, no credits for avoided fertiliser production were accounted for in the model.

The overall composting inventory was implemented in the lifecycle model according to the Circular Footprint Formula, considering it as a case of recycling with a value of the A factor equal to 0.5. Only 50% of the burdens from the composting process were thus allocated to the packaging film life cycle. Conversely, the burdens and benefits from on-land application of the residual composted material were entirely allocated to it.

**Table 5.10.** Biodegradability values considered for biodegradable food packaging films (PLA, starch-based, PLA-based) under aerobic composting conditions.

Polymer	Biodegradation rate (%)	Source
PLA (neat, flexible)	80	UBA (2018) – Amorphous film (70%) Hermann et al. (2011) – Bags (90%)
TPS/PBAT blend	70	UBA (2018) – TPS/PCL (Mater-Bi class Z; 90%) Deconinck and De Wilde (2013) – Starch/PCL blend (90%) Hermann et al. (2011) – different reviewed values for Mater-Bi and Starch/PCL films and bags (27% film, 60%, 75% film, 90% bags)
PLA/PBAT blend	70	UBA (2018) – PLA/PBAT film (35 µm; 40%) Deconinck and De Wilde (2013) – PBAT/PLA blend Ecovio (100%) <sup>(1)</sup>

<sup>(1)</sup> Refers to a duration of 81 days at elevated temperatures.

#### 5.4.5.7 Modelling of anaerobic digestion

Similarly to composting, a waste-specific inventory was developed for anaerobic digestion of biodegradable food packaging films (PLA, Starch-based and PLA-based). The modelling was carried out according to the general recommendations specified in the Method (Report I – Section 4.4.12.5), while relying on relevant process-specific burdens and element-specific transfer coefficients available in the EASETECH model (Clavreul et al., 2014) for the chosen digestion technology. A wet thermophilic digestion process was considered as a reference for modelling purposes, since most biopolymers were found not to degrade (or to degrade only to a very limited extent) under mesophilic conditions, and

<sup>69</sup> As a base case, the effects of non-released biogenic carbon are not reflected in Climate Change LCIA results of biodegradable food packaging films, since the characterisation factors applied to biogenic CO<sub>2</sub> flows were set to zero (fully conforming to provisions in the PEF method). The consequences of this are not significant, since the share of residual, non-emitted carbon is low, equalling 2.1-3.2% of total carbon in the composted polymers. However, to show the relevance of this missed quantification on comparative scenario results, the Climate Change impact of biodegradable packaging film scenarios were also calculated by acknowledging the effects of non-released biogenic carbon. For this purpose, a specific CO<sub>2</sub> uptake was modelled in the composting inventory, corresponding to the amount of biogenic carbon not released over 100 years, which is equal to 0.0271 kg CO<sub>2</sub>/kg PLA film sent to composting, 0.0147 kg CO<sub>2</sub>/kg TPS/PBAT film, and 0.0285 kg CO<sub>2</sub>/kg PLA/PBAT film.

1 most experimental values of anaerobic biodegradability are hence available for  
2 thermophilic conditions (e.g. in UBA, 2018). However, even if thermophilic digestion is  
3 applied, biopolymers degradation under anaerobic conditions is only partial. Therefore,  
4 an additional aerobic post-composting step of the residual, non-digested or partially  
5 digested material in the digestate output was considered, to achieve suitable levels of  
6 biodegradation of the treated bioplastic articles. This additional aerobic step is also  
7 normally applied in most commercial systems for anaerobic digestion of organic waste, to  
8 ensure stabilisation of the biological activity and complete maturation of the digestate  
9 output. The residual organic material from post-composting of digested bioplastic articles  
10 was finally assumed to be applied on agricultural land as soil amendment.

11 The elemental composition considered in the modelling for each packaging film material  
12 is the one already reported in Table 5.8, while the applied biodegradation/mineralisation  
13 rates under anaerobic digestion conditions are summarised in Table 5.11. The latter were  
14 defined as the average of (experimental) values available in the literature for films or  
15 similar plastic articles. Since these values have been at least partly determined under  
16 real digestion or laboratory conditions, they were preferred to the application of the  
17 minimum biodegradation rate required by the European standard for compostable plastic  
18 packaging (EN 13432), i.e. 50% when testing anaerobic treatability. This has indeed to  
19 be proven under optimised laboratory testing conditions, and may not be achieved in real  
20 digestion plants (or it may anyway differ from values obtained from real laboratory  
21 testing available in the literature).

22 Prior to digestion, the incoming waste material was assumed to undergo a pre-treatment  
23 stage, where it is also shredded to make it suitable for further processing in the digestion  
24 plant. In this stage, 30% of biodegradable packaging film collected for treatment was  
25 assumed to be removed as residue, and sent to incineration (modelled as described in  
26 Section 5.4.5.4). The electricity consumption attributed to these pre-treatment  
27 operations was equal to 0.005 kWh/kg input waste.

28 Carbon in biodegradable packaging film sent to digestion was assumed to be converted  
29 (mineralised) into biogas according to the biodegradation rates reported in Table 5.11.  
30 The shares of gasified carbon converted into Methane and Carbon Dioxide were  
31 calculated based on the stoichiometry of the anaerobic degradation reaction, taking into  
32 account the proportions of C, H, O and N in the digested polymers. Carbon conversion  
33 shares are thus material-specific and range from 50% (PLA) to 53% (starch-based  
34 polymer) for Methane, and from 47% (starch-based polymer) to 50% (PLA) for Carbon  
35 Dioxide (Table 5.12). Table 5.12 summarises the parameters relevant to the modelling of  
36 Carbon and VS degradation in the digestion process, some of which are further discussed  
37 below.

38

**Table 5.11.** Biodegradability values considered for biodegradable packaging films (PLA, starch-based, PLA-based) under thermophilic anaerobic digestion conditions.

Polymer	Biodegradation rate (%)	Source
PLA (neat, flexible)	42.5	UBA (2018) – PLA film -0.1 mm- (25%); UBA (2018) – P(L)LA film -30/35% crystalline (60%) <sup>(1)</sup>
TPS/PBAT blend	22	UBA (2018) – TPS/PCL (Mater-Bi) 20 µm film (23%) <sup>(2)</sup> UBA (2018) – Starch/co-polyester (Ecobras) 720 µm film (20%) <sup>(3)</sup>
PLA/PBAT blend	36	Weighted average of the biodegradation rates of PLA (43%) and PBAT (21%) <sup>(4)</sup> , based on the respective content in the polymer blend (70% PLA; 30% PBAT)

<sup>(1)</sup> Refers to a duration of 40 days, which appears long for thermophilic digestion conditions.

<sup>(2)</sup> Refers to mesophilic digestion conditions (35°C), but aligns with the other reported value (which refers to thermophilic conditions and to a duration of 50 days).

<sup>(3)</sup> Refers to a duration of 50 days, which appears long for thermophilic digestion conditions.

<sup>(4)</sup> Assumed to be half of the biodegradation rate of PLA (42.5%), in line with results for composting conditions.

A small share (2%) of the Methane contained in the generated biogas is leaked to air directly at the plant, and represents the only direct emission from the digestion process. Additional process-specific burdens attributed to the latter include those associated with electricity supply for running the process (pumps, ventilators, heating; 0.049 kWh/kg of input waste to digestion), thermal energy usage to heat the reactors (0.192 MJ/kg of input waste), and diesel consumption in plant machinery (0.0009 litre/kg of input waste).

Biogas from digestion (except leakage) is entirely sent to combustion in stationary engines for combined heat and power production. Engines operate with a net electricity efficiency equal to 39%, and a net heat efficiency of 51%. Generated electricity was assumed to replace electricity from the EU grid mix, while generated heat replaced average thermal energy supply at the EU level, considering the same mix of heat sources defined for heat recovered from waste incineration (i.e. 42.4% natural gas, 30.8% hard coal, 21.8% biomass, and 5% heavy fuel oil, based on statistics from the International Energy Agency; IEA, 2019). According to the Circular Footprint Formula, all the burdens and benefits associated with energy generation from biogas were entirely allocated to the system generating the waste sent to anaerobic digestion (i.e. the packaging film life cycle). Emissions from biogas combustion included input-dependent emissions of both fossil and biogenic CO<sub>2</sub> (depending on the origin of carbon in the polymer), as well as process-specific emissions of a number of substances, quantified proportionally to the volume of biogas combusted (e.g. CO, NO<sub>x</sub>, N<sub>2</sub>O, NH<sub>3</sub>, Particulate, metals, NMVOC, and other organic compounds).

The amount of residual, non-digested or partially digested bioplastic material in the digestate output was calculated as the sum of non-degraded volatile solids (VS), as well as of ash and water<sup>70</sup> originally present in the input material to digestion. Residual volatile solids were in turn calculated by assuming a ratio between degraded VS and degraded carbon equal to 1.89 (according to the value applied to organic waste in the EASETECH model).

Aerobic post-composting of such residual material from digestion and subsequent on-land application of the resulting organic material were modelled according to the same approach and assumptions described for direct composting of biodegradable packaging film (Section 5.4.5.6). However, an open (windrow) composting facility was considered in the modelling for this more stabilised organic fraction, assuming the typical average

<sup>70</sup> It is acknowledged that the water content in the digestate output may change compared to the original material sent to digestion due to losses and integration during the process. However, the water content of the investigated biodegradable polymers is equal to zero, so that this simplifying assumption does not affect the calculated amount of residual material from digestion.

carbon and VS biodegradation (mineralisation) rate achieved for generic organic waste in this type of facilities (i.e. 76.4% according to the considered EASETECH dataset). This rate was consistently applied to residual organic material derived from digestion of all types of biodegradable polymers considered for packaging film manufacturing, regardless of the level of biodegradation achieved in the first anaerobic digestion step. Based on transfer coefficients (emission factors) reported in the mentioned windrow composting dataset, 97.76% of mineralised carbon was assumed to be converted to CO<sub>2</sub>, and the remaining 2.24% to CH<sub>4</sub>. A final screening stage of composted material was also included, and the respective burdens were accounted in the inventory. These are quantified along with the overall process-specific burdens attributed to the composting process and include diesel combustion in machinery (screens and loaders; 0.003 l/kg input waste), electricity (for services, 0.72 MJ/kg input waste), lubricating grease (0.013 l/kg input waste), motor oil and hydraulic oil (0.01 l/kg input waste), and Carbon Monoxide emissions to air (0.00031 kg/kg input waste).

**Table 5.12.** Parameters applied in the modelling of carbon and VS degradation in the anaerobic digestion process of biodegradable food packaging films.

Parameter	Value	Source
Bio-gasified (mineralised) Carbon	Material-specific	See Table 5.10
C degraded to CH <sub>4</sub>	PLA: 50% TPS/PBAT: 53% PLA/PBAT: 51%	Based on stoichiometry of the anaerobic degradation reaction and content of C, H, O, and N in the specific material
C degraded to CO <sub>2</sub>	PLA: 50% TPS/PBAT: 47% PLA/PBAT: 49%	
CH <sub>4</sub> leakage in the digestion plant	2% of CH <sub>4</sub> produced	Process-specific emission factor (from the EASETECH model)
Degraded VS/Degraded C	1.89	Default value for organic waste (from the EASETECH model)
Residual (non-digested or partially digested) bioplastic material in the output digestate	Non-degraded VS + initial ash & water in the material	Assumption

#### 5.4.5.8 Contribution to macro and micro-plastics generation

The contribution to macro- and micro-plastics generation of the analysed food packaging films scenarios was estimated according to the framework outlined in Annex B of the Method (Report I), and summarised in Section 4.4.5.6 of this report in terms of underlying quantification methods and approaches. In this section, the case-specific parameters considered to apply the latter to the assessed food packaging films LCA scenarios are reported.

##### 5.4.5.8.1 PLP method

Regarding the estimate of the loss and release of macro-plastics at the End of Life stage (from product littering and waste mismanagement), the product-specific parameters reported in Table 5.13 were considered to apply the first level of the framework ("*PLP method*") to food packaging films (derived from Peano et al., 2020). Note that such parameters are defined regardless of the type of feedstock or material used for films manufacturing. Similarly, all food packaging films scenarios contribute to macro-plastics

loss and release to the same extent, as the chosen feedstock or material do not affect the probability of the article to be littered.

**Table 5.13.** Case-specific parameters of the PLP method applied to the food packaging films LCA scenarios.

Parameter <sup>(1)</sup>	Value
Littering rate (LR <sub>lit</sub> ) (%)	5
Release rate to ocean (Rel <sub>ocean</sub> ) (%)	25
Release rate to the terrestrial environment (Rel <sub>terenv</sub> ) (%)	75

<sup>(1)</sup> For details on the meaning of each parameter, refer to Annex B of the Method (Report I).

No case-specific parameters are required to calculate the supply-chain loss and release of micro-plastics to ocean and to the terrestrial environment by applying the *PLP method*. Indeed, the contribution of the considered micro-plastics sources (pellet loss and tire abrasion) depends on the magnitude of the relevant life cycle processes (i.e. polymer production and road transport) in the specific food packaging films scenario (which in turns depend on the reference flow<sup>71</sup> of each scenario). However, the quantification is made by means of default parameters that are not affected by the type of product, polymer or feedstock source (as reported in Annex B of the Method (Report I)).

#### 5.4.5.8.2 Expanded PLP method

The parameters required to calculate the micro-plastics loss and release from marine coatings and road markings are not case-specific, and are reported in Annex B of the Method (Report I). Similar to the other considered micro-plastics sources, the contribution of these additional sources depends on the intensity of relevant (transport) processes in the specific product inventory, by means of the above-mentioned case-unspecific parameters.

As for the calculation of the additional contribution of plastic waste exported for recycling to the loss and release of macro-plastics from waste mismanagement, the case-unspecific parameters described in Annex B of the Method were applied, in combination with the EU-average recycling rates estimated for each type of food packaging films (Section 5.4.5.1). For the sake of clarity, the applied recycling rates are summarised again in Table 5.14. Note that bio-based scenarios are considered to be composted instead of following a recycling path.

**Table 5.14.** EU-average recycling rates considered for each food packaging films scenario to estimate the amount of waste plastic films exported to non-EU countries for recycling.

Polymer	Recycling rate
PP	20%
LDPE	
Bio-LDPE	
PLA	0%
PLA-based	
Starch-based	
CO <sub>2</sub> -based PP	20%

<sup>71</sup> The reference flow is the mass of film material required to fulfil the functional unit.



#### 5.4.5.8.3 Sensitivity analysis 1: alternative calculation of the total plastic loss along the supply-chain

This sensitivity analysis was not applied to the packaging food films case study.

#### 5.4.5.8.4 Sensitivity analysis 2: alternative calculation of macro-plastics release to ocean

The EU-average marine litter rate (MLR, %) for food packaging films and the resulting release to ocean (kg) were estimated based on beach litter observations and apparent consumption data at the EU level, according to the procedure reported in Annex B of the Method (Report I). For calculation purposes, overall values for the period 2012-2016 were considered for both quantities. Moreover, the following product categories were considered in the respective data sources, reported as well below:

- Beach litter observations (Beach litter database; Addamo et al., 2018; Hanke et al., 2020): "*Crisps packets, sweets wrappers, lolly sticks*" (B18), as representative use of food packaging films.
- EU Apparent consumption (PRODCOM database; Eurostat, 2019c): "*Articles for the conveyance or packaging of goods, of plastics (excluding boxes, cases, crates and similar articles; sacks and bags, including cones*", since no specific category for packaging food films was available.

Table 5.15 summarises the values considered in the calculation, along with the obtained estimates of the beach litter rate and of the resulting (up-scaled) marine litter rate.

**Table 5.15.** Estimate of the beach litter rate and of the resulting up-scaled marine litter rate for food packaging films.

<b>Total predicted beach litter <sup>(1)</sup> (kg/year)</b>	<b>Total apparent EU consumption <sup>(2)</sup> (kg/year)</b>	<b>Beach Litter rate (%)</b>	<b>Marine Litter rate (%)</b>
1.77E+05 <sup>(3)</sup>	4.10E+09	0.0043	0.0864

<sup>(1)</sup> Along the entire length of EU coast. Estimated based on observed beach litter data over 100 m of coastline (from Addamo et al., 2018 and Hanke et al., 2020).

<sup>(2)</sup> From PRODCOM (Eurostat, 2019c).

<sup>(3)</sup> Assumed average weight = 1.14g per unit.

### 5.4.6 Calculation of iLUC impacts

As a base case, the contribution of Indirect Land Use Change (iLUC) to the potential Climate Change impact of bio-based or partially bio-based food packaging film alternatives was calculated according to the approach outlined in Section 4.4.16.4 of the Method (Report I). A sensitivity analysis on the applied method has been performed, with the respective results being presented in section 5.7.7.7.

In order to apply (recalculated) iLUC GHG emission factors from the EU 2015/1513 Directive (EC, 2015), the specific land demand for the production of relevant crops (m<sup>2</sup>\*year / kg crop) was calculated first. The calculation was based on the aggregated amount of arable and agriculture land occupation flows reported, for the relevant country (e.g. Brazil for sugarcane), in the dataset used to model the production of the specific crop. If the geography of such flows was not specified, all the arable and agricultural land occupation flows reported in the dataset were aggregated. These estimates were checked against values of land demand calculated based on 5-years average crop yields from FAOSTAT (FAO, 2019), which were found to be generally aligned with the former (absolute variation between 3% and 20%). Hence, the values estimated based on land occupation flows were applied, for consistency with the overall LCI modelling of scenarios.

Specific land demand for crop production was then converted into a demand per functional unit, based on the specific consumption of crop for polymer production (kg crop/kg polymer, consistently with the values applied in the LCI modelling) and the

- 1 amount of polymer needed to fulfil the functional unit (reference flow) in the relevant
- 2 scenario. The iLUC contribution to the Climate Change impact was finally calculated by
- 3 applying to the latter the recalculated GHG emission factors from the Directive. The
- 4 described calculation steps to estimate the iLUC contribution to the potential Climate
- 5 Change impact are summarised in Table 5.16.

**Table 5.16.** Calculation of the iLUC contribution to the potential Climate Change impact of food packaging film LCA scenarios relying on bio-based polymers.

Scenario / Polymer	Feedstock	Land demand for crop production <sup>(1)</sup> [m <sup>2</sup> *y/kg <sub>crop</sub> ]	Crop demand for polymer production [kg <sub>crop</sub> /kg <sub>polymer</sub> ]	Polymer demand per F.U. [kg <sub>polymer</sub> /FU]	iLUC GHG emission factor [kg CO <sub>2</sub> eq/(m <sup>2</sup> *y)]	iLUC climate change impact [kg CO <sub>2</sub> eq/FU]
S3 – Bio-LDPE	Sugarcane (BR)	0.117 (0.135)	23.8	5.33	0.176	2.61
S4 – PLA	Maize (US)	1.06 (0.939)	3.08	1.67	0.0612	0.334
S5 – TPS/PBAT	Maize (EU)	1.34 (1.37)	0.134	3.18	0.0612	0.0348
	Wheat (EU)	1.52 (1.75)	0.121	3.18	0.0612	0.0359
	Potatoes (EU)	0.376 (0.301)	0.140	3.18	0.0612	0.0103
	Total					0.0810
S6 – PLA/PBAT	Maize (US)	1.06 (0.939)	2.16	4.25	0.0612	0.597

<sup>(1)</sup> Based on arable and agriculture land occupation exchanges reported in the dataset applied to the modelling of the specific crop. Values in parenthesis refer to land demand calculated based on crop yield data from FAOSTAT (5-years average), and are reported as a reference.

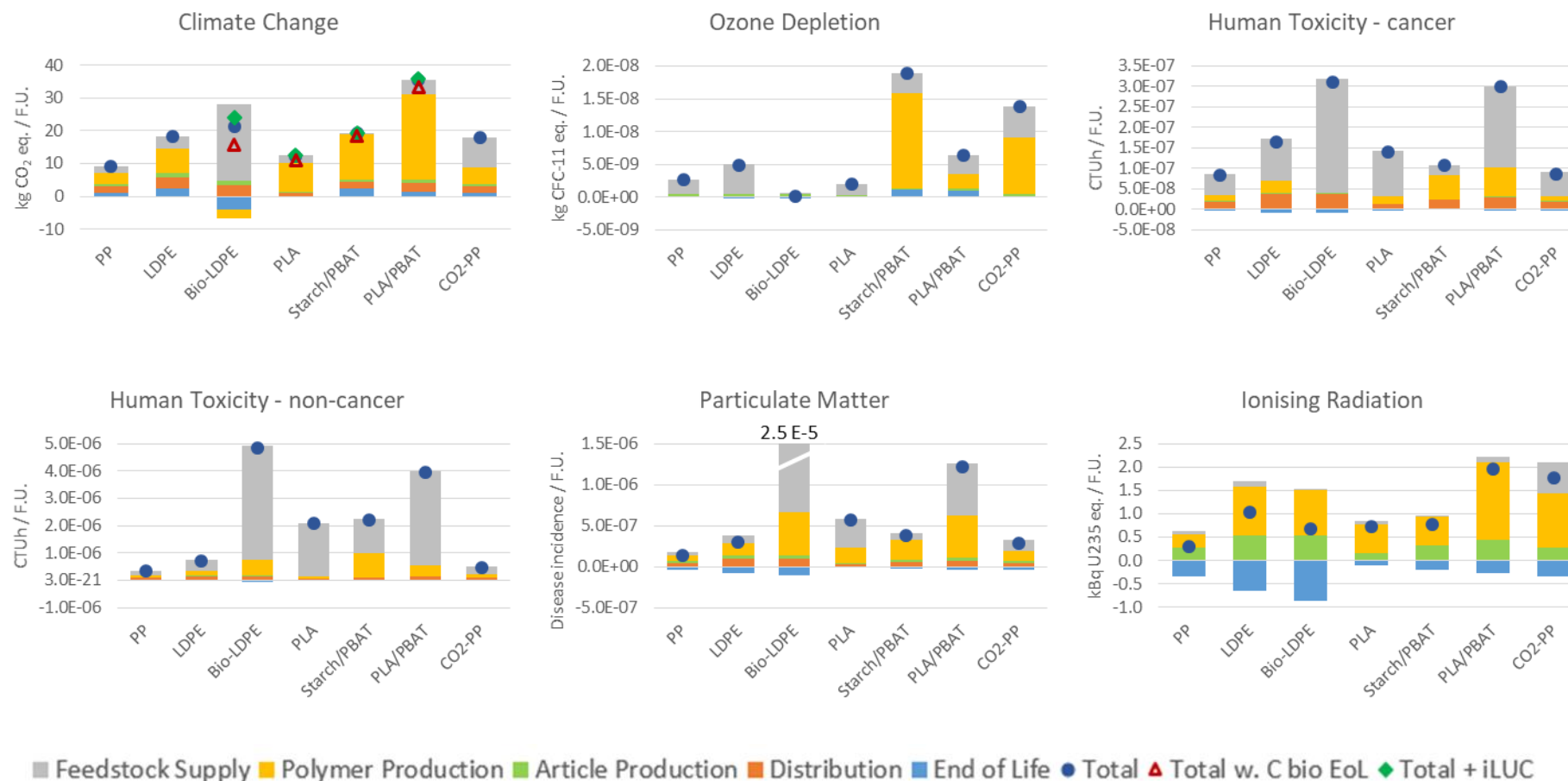
## 5.5 Life Cycle Impact Assessment results

The characterised potential impacts of the examined scenarios (with and without the iLUC contribution to Climate Change) are reported in Figures 5.8 to 5.10. These also show the breakdown of contributions from the main lifecycle stages, which include:

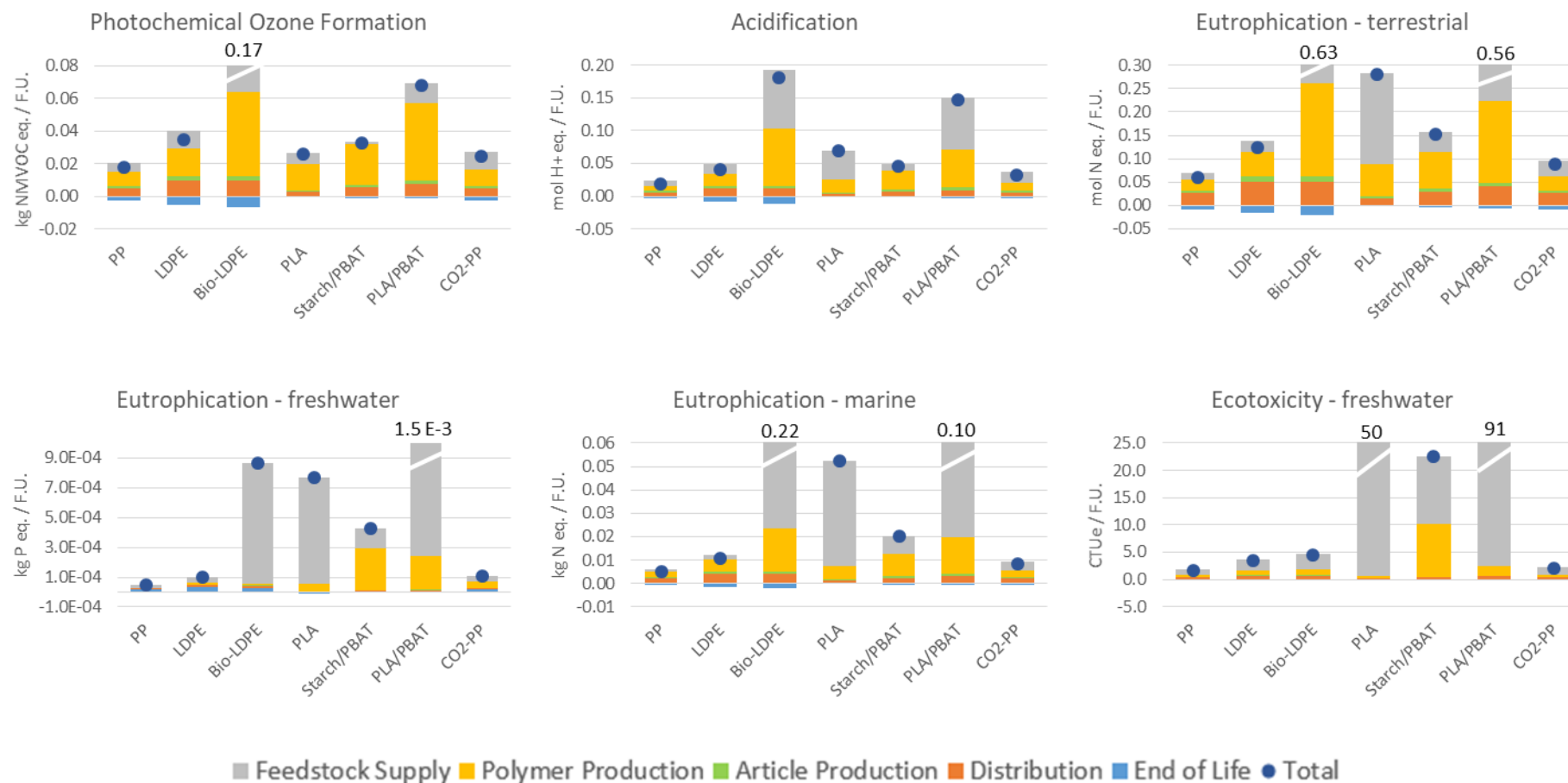
- i) *Feedstock Supply*, i.e. oil/natural gas extraction, transport and possible refining, transport of naphtha from refinery to downstream users; crop cultivation and transport to further processing (bio-based polymers); or CO<sub>2</sub> capture and Hydrogen production for subsequent utilisation and their transport via pipeline;
- ii) *Polymer Production*, i.e. all gate-to-gate activities from feedstock processing to monomer/intermediate production and polymerisation, including any transport among these, as well as transport of polymer granulate to the packaging film manufacturing site;
- iii) *Article Production*, i.e. conversion of the polymer into packaging film by blown film extrusion and subsequent transport to downstream users (i.e. producers of packaged food);
- iv) *Distribution*, i.e. transport of packaging film from the food packing site to the final user, as a defined packaging item (e.g. bag) or component (e.g. lid or wrap); and
- v) *End of Life*, i.e. waste packaging film collection, transport and treatment or disposal, as well as any avoided processes from downstream displacement of virgin materials and energy).

Figure 5.11 shows the contribution of each sub-category to the Climate Change impact (i.e. from fossil, biogenic and land use/land transformation GHG emissions), while the overall characterised, normalised and weighted impact assessment results are available in Annex B.2. The latter are not reported in this section to keep readability of the document, despite this is not in alignment with the position suggested in the LCA report template provided in the Method (Report I).

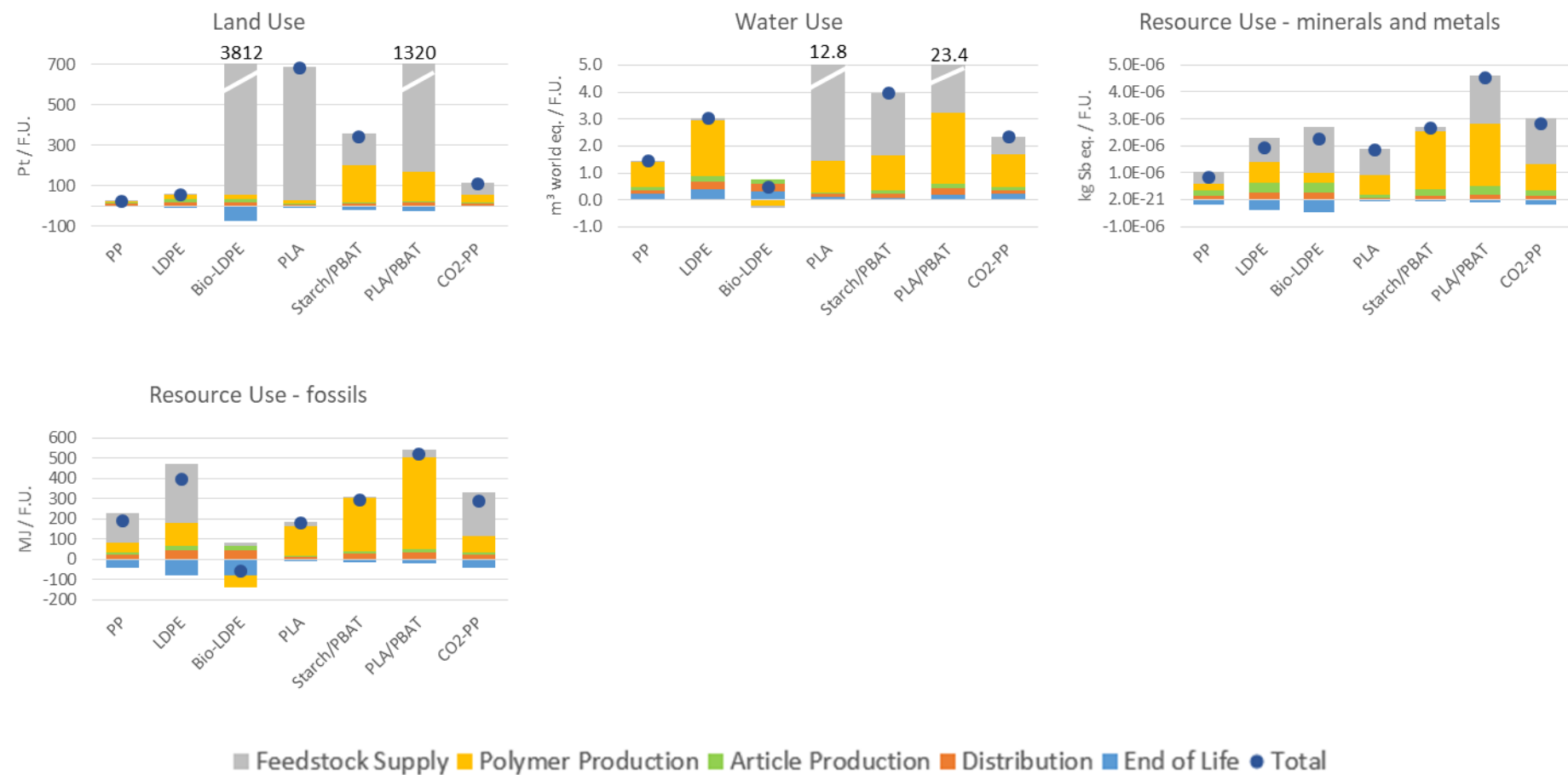
Note that scenario impacts presented in Figures 5.8 to 5.10 refer to the EU-average End of Life scenario (as described in Section 5.4.5.1), and represent the net impacts from waste management activities, resulting from the balance between real burdens and benefits (if any). Potential impacts calculated assuming 100% of post-consumer food packaging film being routed to each viable End of Life option are presented in Figures 5.12 to 5.14, except for Bio-LDPE and PLA/PBAT food packaging films, which for reasons of scale are separately presented in Annex B.2. We warn that this analysis should not be interpreted as a direct LCA comparison of alternative End of Life management options for food packaging film, because a product-based perspective is considered, and allocations among consecutive product life cycles are applied to the recycling pathway. Conversely, in a typical waste management LCA that compares alternative End of Life options (i.e. with a functional unit of 1 tonne of waste managed), the recycling pathway would be assigned the full burdens and benefits, as there is no need to break the system mass flows between the system recycling and the one using the recycled feedstock (i.e. system perspective; no allocation needed).



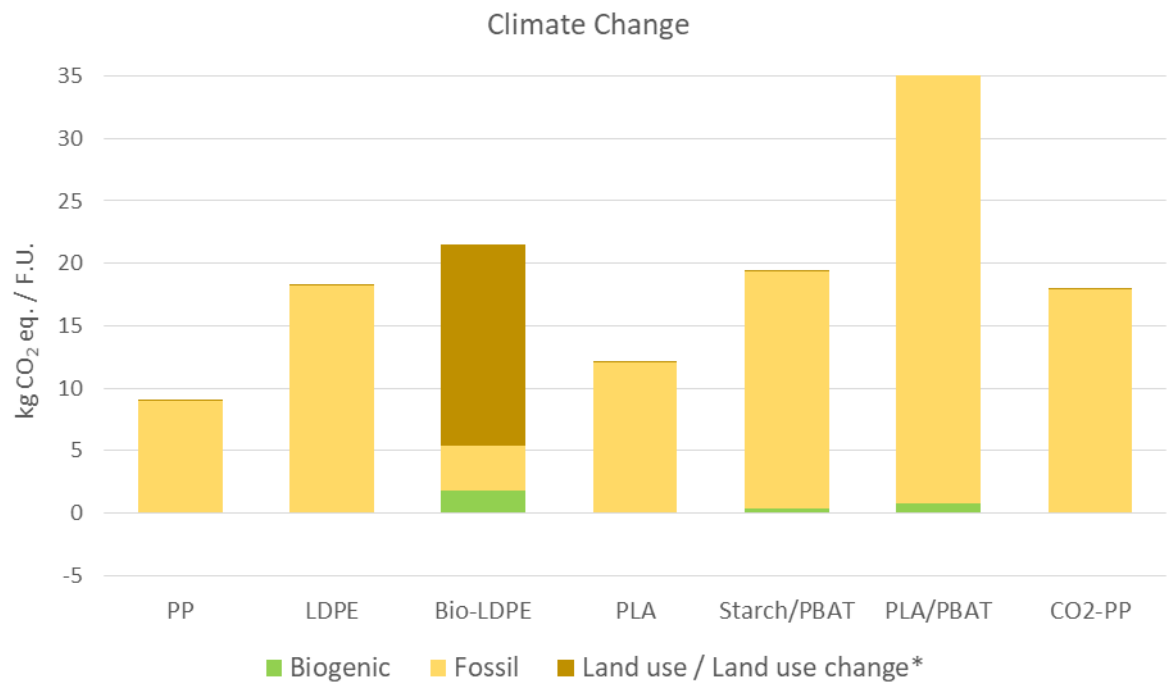
**Figure 5.8.** Potential impact of food packaging film LCA scenarios for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity – non-cancer, Particulate Matter and Ionising Radiation. Note that in some impact categories a part of the results is out of scale and is curtailed. Climate Change impacts denoted with “C bio EoL” accounts for the contribution of biogenic carbon not released after 100 years from landfilling of bio-based food packaging films or from on-land application of residual organic matter derived from their composting or anaerobic digestion.



**Figure 5.9.** Potential impact of food packaging film LCA scenarios for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater. Note that in some impact categories a part of the results is out of scale and is curtailed.



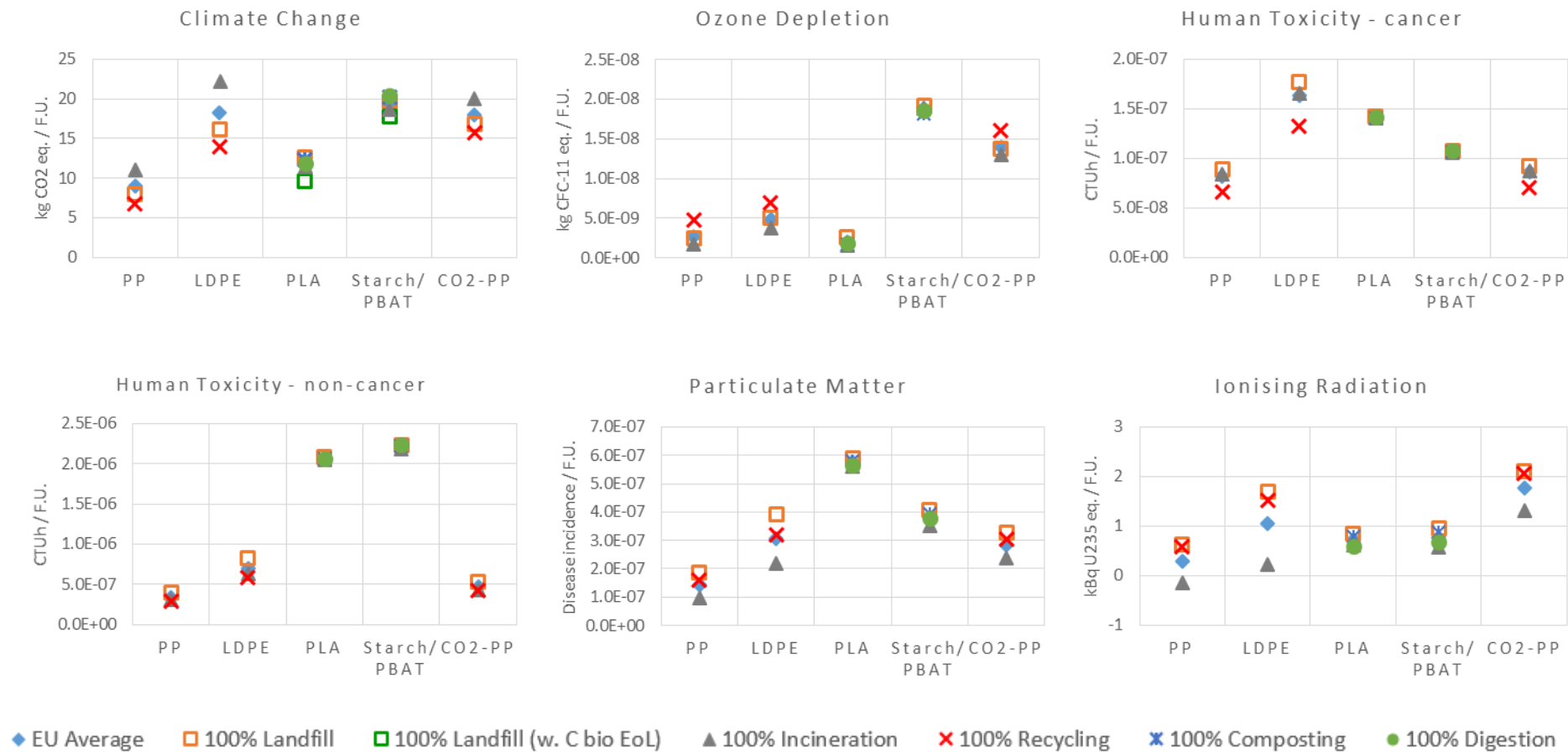
1 **Figure 5.10.** Potential impact of food packaging film LCA scenarios for the categories of Land Use, Water Use, Resource Use - minerals and metals,  
2 Resource Use – fossils. Note that in some impact categories a part of the results is out of scale and is curtailed.



**Figure 5.11.** Contribution of fossil, biogenic, and land use/land transformation GHG emissions to the total Climate Change impact of food packaging film LCA scenarios (note that the contribution of iLUC to land transformation emissions is excluded).

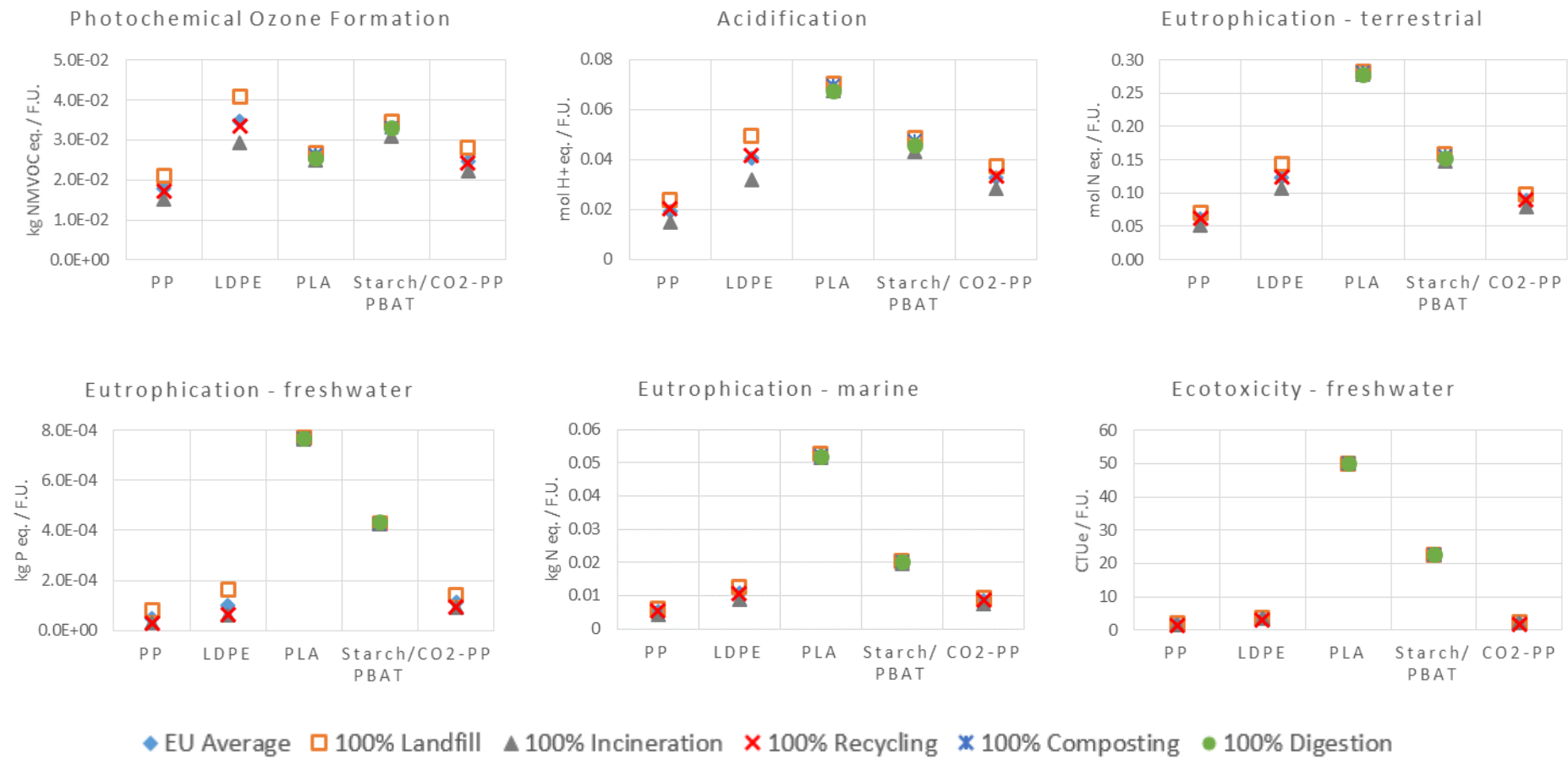


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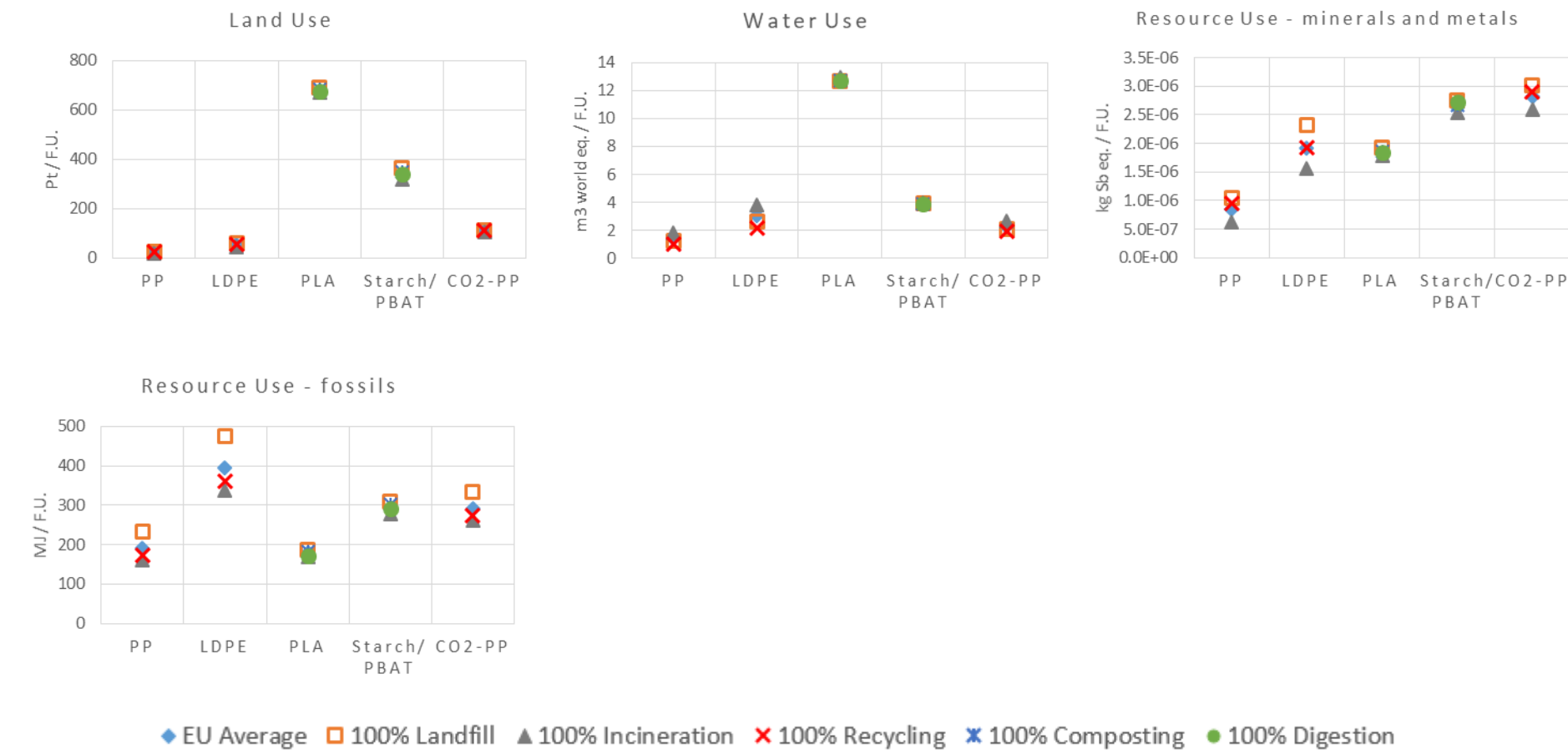


**Figure 5.12.** Potential impact of food packaging film LCA scenarios for different End of Life options, for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity – non-cancer, Particulate Matter and Ionising Radiation. The Climate Change impact of 100% landfilling scenario denoted with “C bio EoL” accounts for the contribution of biogenic carbon not released after 100 years from landfill deposition of bio-based food packaging films.

1



**Figure 5.13.** Potential impact of food packaging film LCA scenarios for different End of Life options, for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater.



**Figure 5.14.** Potential impact of food packaging film LCA scenarios for different End of Life options, for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils.

## 5.6 Additional Environmental Information

This section presents additional environmental impacts or aspects going beyond the default impact categories considered in the present method, which reflect official categories adopted in the PEF context. Additional environmental issues addressed include the impact from indirect Land Use Change (iLUC) on Climate Change, potential Biodiversity impacts, as well as the contribution of the investigated food packaging film scenarios to macro-plastics formation at End of Life (including product litter) and to micro-plastics generation throughout the supply chain.

### 5.6.1 iLUC impact

Table 5.17 presents the estimated contribution of GHG emissions from iLUC induced by bio-based feedstock supply to the potential Climate Change impact of bio-based food packaging films.

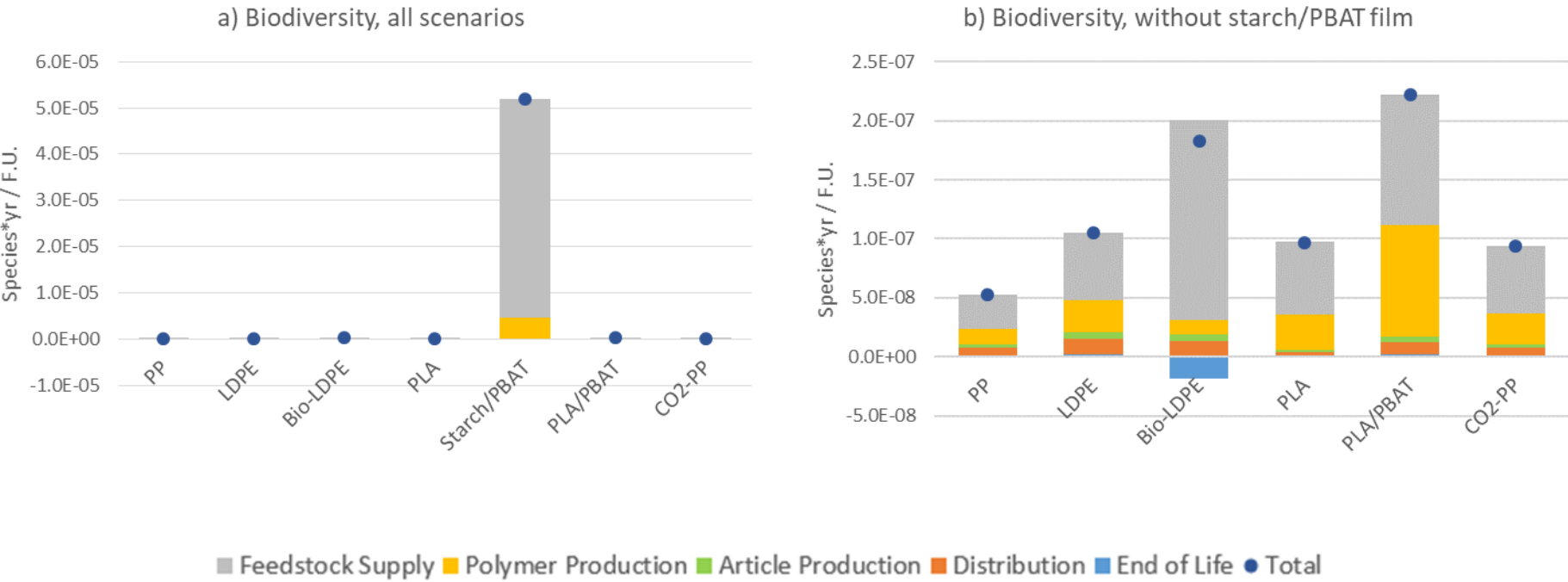
**Table 5.17.** iLUC contribution to the potential Climate Change impact of packaging film LCA scenarios.

Scenario / Polymer	iLUC Climate Change impact [kg CO <sub>2</sub> eq/FU]	Total Climate Change impact (incl. iLUC) <sup>(1)</sup> [kg CO <sub>2</sub> eq/FU]
S1 – PP	-	(9.0)
S2 – LDPE	-	(18.2)
S3 – Bio-LDPE	2.6	21.5 (24.1)
S4 – PLA	0.3	12.1 (12.4)
S5 – Starch/PBAT	0.1	19.3 (19.4)
S6 – PLA/PBAT	0.6	35.4 (36.0)
S7 – CO <sub>2</sub> -based PP	-	(17.9)

<sup>(1)</sup> Values in parenthesis refer to the Climate Change impact without the iLUC contribution.

### 5.6.2 Biodiversity impacts

Estimated potential Biodiversity impacts of the investigated scenarios are presented in Figure 5.15. Impacts are separately presented by including and excluding the results related to Starch-based food packaging film, to allow appreciating differences among the other compared scenarios. It is reminded that the assessment is performed through an endpoint-level indicator relying on impact assessment methods that for some of the underlying midpoint impact drivers differ from those recommended in the present method.

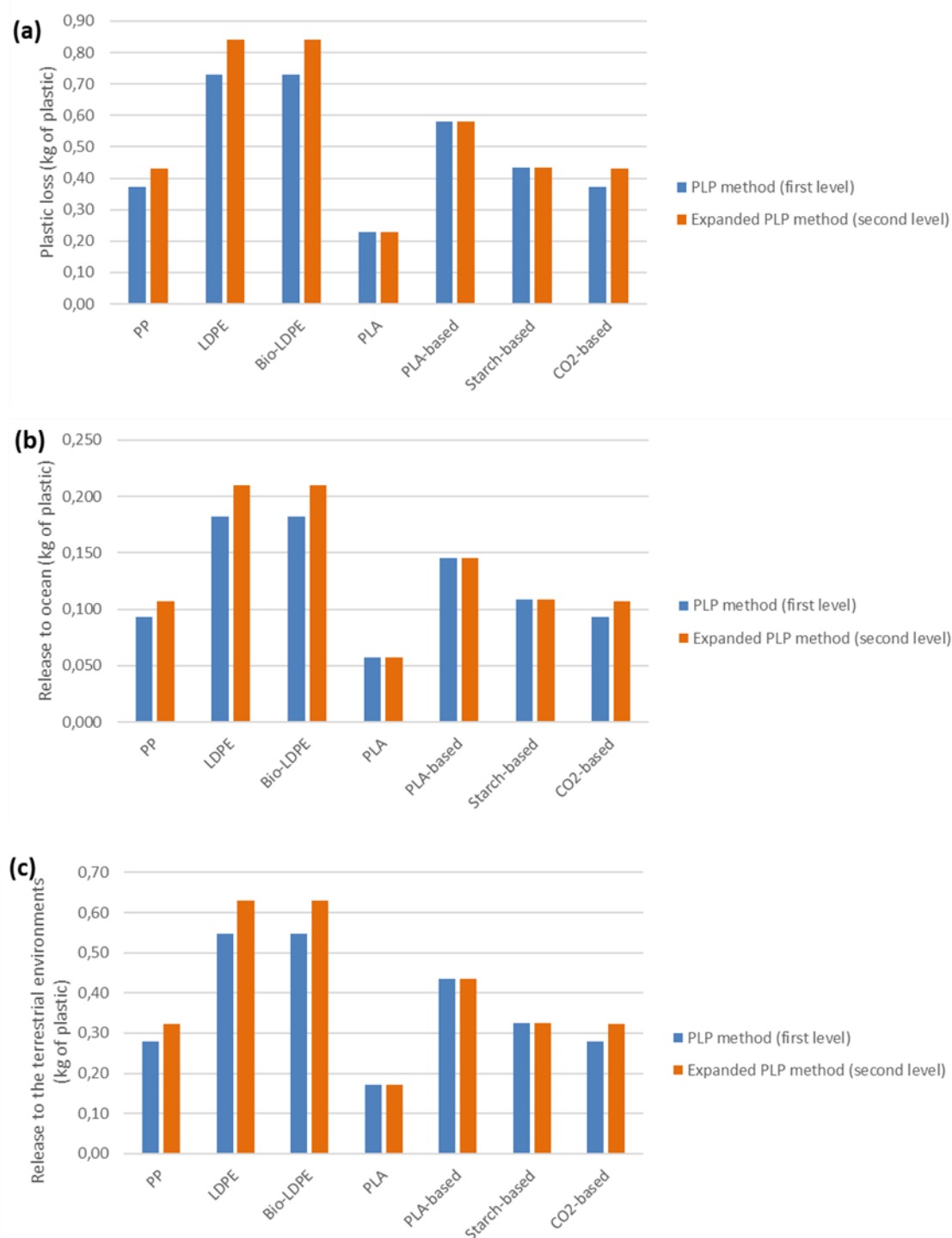


**Figure 5.15.** Potential biodiversity impact of food packaging film LCA scenarios, expressed as potential loss of species per year, including (a) and excluding (b) starch-based food packaging film.

### **5.6.3 Macro- and micro-plastics generation**

Figure 5.16 shows the total plastic loss, release to ocean and release to the terrestrial environment estimated for the assessed food packaging film scenarios, considering the whole product life cycle. Estimates obtained by applying all the two different explored approaches are presented, including the total loss and release to ocean/terrestrial environment calculated through both the PLP method (first level) and the Expanded PLP method (second level).

1

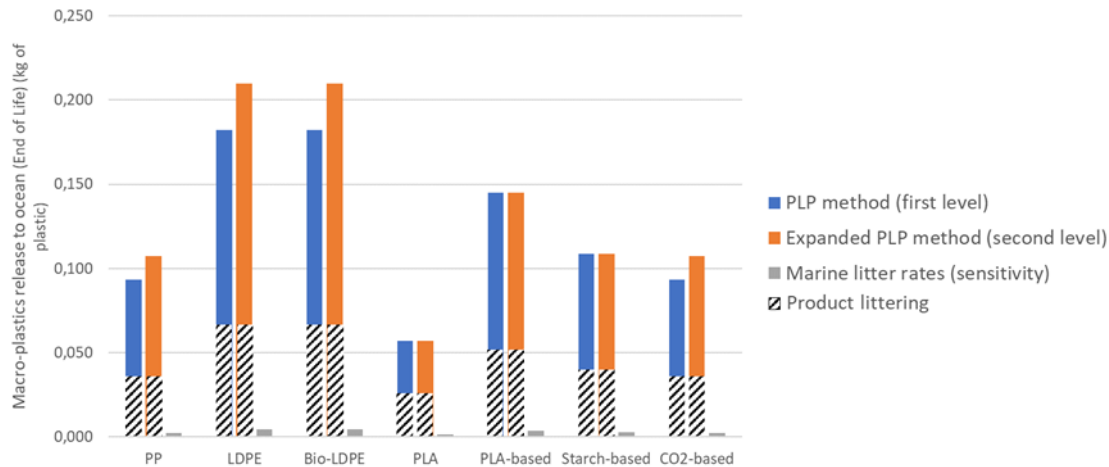


2

**Figure 5.16.** Total plastic loss (a), release to ocean (b) and release to the terrestrial environment (c) estimated for food packaging films LCA scenarios with different approaches. The contribution of both macro- and micro-plastics is included, considering the whole product life cycle.

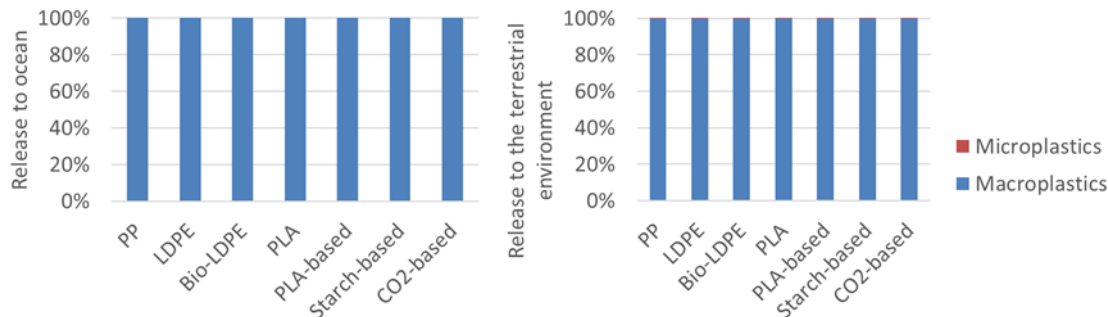
Regarding the sole release of macro-plastics to ocean at End of Life (from product littering and waste mismanagement), Figure 5.17 presents the contribution of each food packaging films scenario, estimated with the three different explored approaches: the PLP method (first level), the Expanded PLP method (second level), and the sensitivity approach based on product-specific marine litter rates from observed beach litter in the EU. Note that the latter do not differentiate the type of source (e.g. littering and waste

mismanagement), but inherently takes into account all the possible sources that may ultimately contribute to macro-plastic release to ocean at the European level (still with a higher degree of uncertainty).



**Figure 5.17.** Macro-plastics release to ocean at End of Life, estimated with different approaches.

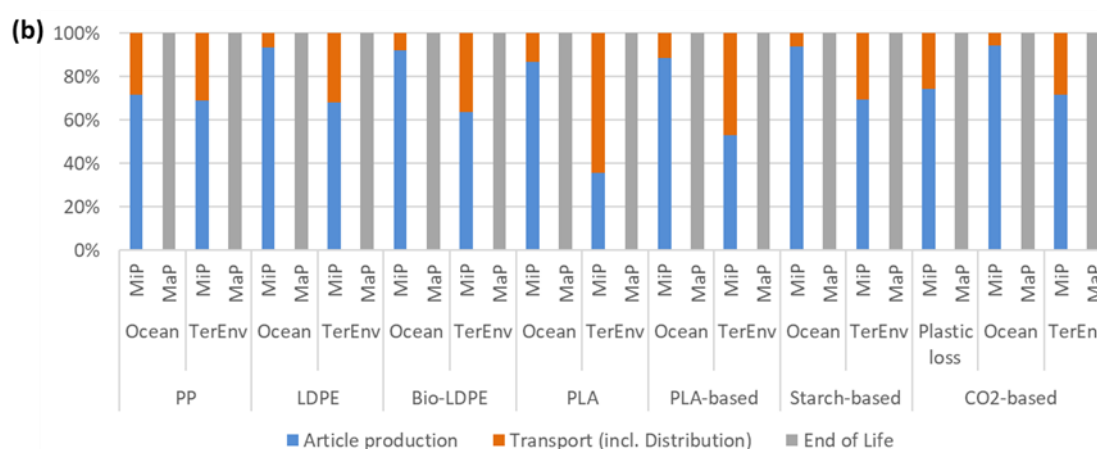
As for the share between the release of macro- and micro-plastics, Figure 5.18 shows as an example the relative contribution of the two categories to the total release to ocean and to the terrestrial environment estimated through the PLP method (first level). Note that the results are similar to those obtained when applying the Expanded PLP method (second level), as further discussed in Section 5.7.7.



**Figure 5.18.** Contribution of macro- and micro-plastics to the total plastic release to ocean and to the terrestrial environment estimated with the PLP method for each food packaging films LCA scenario.

The contribution of the different lifecycle stages to the total macro- and micro-plastics release to ocean and to the terrestrial environment of the assessed food packaging films scenarios has been finally evaluated. The results of this “hotspot analysis” are reported in Figure 5.19 for both the PLP method and the Expanded PLP method. Results are not shown for the other sensitivity approach as it covers only a part of the product life cycle (i.e. End of Life).





**Figure 5.19.** Contribution of life cycle stages to the total macro-plastics (MaP) and micro-plastics (MiP) release to ocean (Ocean) and to the terrestrial environment (TerEnv) estimated for food packaging films LCA scenarios with (a) the PLP method (first level) and (b) the Expanded PLP method (second level).

## 5.7 Interpretation

In the interpretation of case study results, most relevant impact categories and lifecycle stages of the studied systems are firstly identified (5.7.1 and 5.7.2, respectively). Scenario impacts are then compared, considering both characterised LCIA results (5.7.3), and total normalised and weighted impacts (5.7.4). Results obtained by individually applying each viable End of Life option are separately discussed in Section 5.7.6, while the effects of iLUC and the contribution to macro- and micro-plastics generation are addressed in Sections 5.7.5 and 5.7.7, respectively. Finally, the results of the sensitivity analysis on relevant parameters and assumptions are presented (5.7.8).

Most relevant processes were not identified, since it was not possible to achieve a similar level of vertical disaggregation for foreground processes included in the life cycle models of the analysed scenarios (e.g. a higher disaggregation could be achieved for Starch-based and CO<sub>2</sub>-based PP packaging films, but this was not the case of other scenarios). Therefore, the identification of most relevant processes cannot be carried out consistently across all scenarios, and a more detailed investigation for specific scenarios would not be meaningful. The identification of most relevant elementary flows was also not undertaken, as this would require prior identification of most relevant processes. Note, however, that any company, organisation or any other supply chain actor applying the

present method shall proceed with the identification of both most relevant processes and elementary flows.

### **5.7.1 Identification of most relevant impact categories**

Table 5.18 shows the most relevant impact categories identified for each food packaging film scenario, based on normalised and weighted impacts, according to the approach outlined in the Method (Report I, Section 6.2.1). Relevant categories were identified as those that cumulatively contribute to at least 80% of the total normalised and weighted impact of the considered impact categories, excluding toxicity-related ones (these being still based on characterisation factors implemented in the EF 2.0 impact assessment methods, and hence not yet updated based on REACH data<sup>72</sup>). 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 fossil-based and CO<sub>2</sub>-based PP food packaging film, as well as fossil-based LDPE film, three impact categories are identified as the most relevant, although only two of them are responsible for more than 80% of the total impact, i.e. Climate Change and Resource Use – fossils. Climate Change is the most relevant category for CO<sub>2</sub>-based PP film, while for fossil-based PP and LDPE films, Resource use – fossil is the most relevant one. These two categories are also the most relevant for PLA, Starch/PBAT and PLA/PBAT packaging films, although for these alternatives also other impact categories are identified as relevant, still with a lower contribution (i.e. Particulate Matter, Water Use, Acidification and Eutrophication – terrestrial, depending on the polymer). In the case of bio-based LDPE packaging film, Particulate Matter is the most relevant category, followed by Climate Change (and beyond the 80% threshold, by Eutrophication – marine).

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<sup>72</sup> According to the latest version of the Product Environmental Footprint Category Rules Guidance (EC, 2018), toxicity-related impact indicators based on these characterization factors shall be excluded from the procedure to identify most relevant impact categories.

**Table 5.18.** Most relevant impact categories identified for food packaging film LCA scenarios and related contribution to the total normalised and weighted scenario impact.

S1 - Fossil-based PP		S2 – Fossil-based LDPE		S3- Bio-based LDPE	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Resource Use – fossils	41.9%	Resource Use - fossils	42.1%	Particulate Matter	70.0%
Climate Change	41.4%	Climate Change	40.7%	Climate Change	11.4%
Acidification	3.7%	Acidification	3.8%	Eutrophication - marine	4.5%
<b>Total</b>	<b>87.0%</b>	<b>Total</b>	<b>86.5%</b>	<b>Total</b>	<b>85.9%</b>
S4 – PLA		S5 – Starch/PBAT blend		S6 – PLA/PBAT blend	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Climate Change	32.2%	Climate Change	44.8%	Climate Change	37.2%
Resource Use – fossils	22.7%	Resource Use - fossils	32.6%	Resource Use – fossils	26.2%
Water Use	9.4%	Particulate Matter	4.6%	Water Use	6.8%
Particulate Matter	8.0%			Particulate Matter	6.7%
Acidification	7.6%			Acidification	6.5%
Eutrophication - terrestrial	5.8%				
<b>Total</b>	<b>85.7%</b>	<b>Total</b>	<b>82.0%</b>	<b>Total</b>	<b>83.4%</b>
S7 – CO <sub>2</sub> -based PP					
<i>Impact category</i>	<i>Contrib.</i>				
Climate Change	46.4%				
Resource Use - fossils	36.0%				
Particulate Matter	3.9%				
<b>Total</b>	<b>86.2%</b>				

### 5.7.2 Identification of most relevant life-cycle stages

Table 5.19 shows the most relevant lifecycle stages in the relevant impact categories identified in Section 5.7.1, and the associated contribution, quantified according to the rules detailed in the Method (Report I, Section 6.2.2. Most relevant stages include those that together contribute to at least 80% of the total characterised lifecycle impact in the specific category, and are highlighted in yellow in Table 5.19. For a more exhaustive picture, the contribution of all lifecycle stages is reported.

For fossil-based PP and LDPE food packaging films, Feedstock Supply is identified as the most relevant stage in Resource Use – fossils, followed by Polymer Production. Conversely, Polymer Production is the most relevant stage in the Climate Change

category, followed by Feedstock Supply, Distribution and End of Life. The same order applies as well to Acidification, which in the case of PP film includes also Distribution as a relevant stage.

For CO<sub>2</sub>-based PP packaging film, Feedstock Supply and Polymer production are, in the order, the most relevant stages in all the impact categories identified as most relevant. In the case of Climate Change, also Distribution is included, with a lower contribution, as a third most relevant stage.

For bio-based LDPE packaging film, Feedstock Supply is the most relevant stage in all the identified most relevant impact categories, with a contribution larger than 95% to Particulate Matter, and reaching almost 90% for Climate Change and Eutrophication – marine. This likely reflects the relatively high specific consumption of sugarcane for the production of the polymer (approximately 24 kg of sugarcane per kg of Bio-LDPE).

In the case of PLA and PLA/PBAT packaging films, either Polymer Production (Climate Change and Resource Use – fossils) or Feedstock Supply (Water Use, Particulate Matter, Acidification, and Eutrophication – terrestrial) is identified as the most relevant stage (alone or followed by another stage, depending on the category, i.e. Feedstock Supply or Polymer Production). For starch/PBAT film, Polymer Production is the most relevant stage in all the most relevant categories, followed by End of Life (Climate Change) or Feedstock Supply (Particulate Matter), while in Resource Use – fossils Polymer Production contributes, alone, to nearly 90% of the total impact.

**Table 5.19.** Contribution of life cycle stages to the total characterised impacts of the most relevant categories identified for food packaging film LCA scenarios. Most relevant stages (i.e. those contributing to more than 80% of the total impact) are highlighted in yellow.

<b>S1 – Fossil-based PP</b>					
<b>Resource Use - fossils</b>		<b>Climate Change</b>		<b>Acidification</b>	
<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	77.7%	Polymer production	40.3%	Polymer production	41.1%
Polymer production	26.2%	Feedstock Supply	19.6%	Feedstock Supply	36.8%
Distribution	11.9%	Distribution	19.3%	Distribution	31.2%
Article Production	5.9%	End of Life	13.2%	Article Production	11.6%
End of Life	-21.6%	Article Production	7.6%	End of Life	-20.7%
<b>S2 – Fossil-based LDPE</b>					
<b>Resource Use - fossils</b>		<b>Climate Change</b>		<b>Acidification</b>	
<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	74.7%	Polymer production	41.8%	Polymer production	45.4%
Polymer Production	28.6%	Feedstock Supply	19.3%	Feedstock Supply	34.8%
Distribution	11.2%	Distribution	18.7%	Distribution	28.9%
Article Production	5.6%	End of Life	12.9%	Article Production	10.7%
End of Life	-20.1%	Article Production	7.4%	End of Life	-19.8%

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<b>S3 – Bio-based LDPE</b>					
<b>Particulate Matter</b>		<b>Climate Change</b>		<b>Eutrophication - marine</b>	
<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	97.8%	Feedstock Supply	88.4%	Feedstock Supply	90.4%
Polymer Production	2.1%	Distribution	12.9%	Polymer production	8.3%
Distribution	0.4%	Polymer production	9.3%	Distribution	1.9%
Article Production	0.2%	Article Production	5.1%	Article Production	0.4%
End of Life	-0.4%	End of Life	-15.7%	End of Life	-1.0%
<b>S4 – PLA</b>					
<b>Climate Change</b>		<b>Resource Use - fossils</b>		<b>Water Use</b>	
<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	71.8%	Polymer Production	80.8%	Feedstock Supply	88.5%
Feedstock Supply	19.7%	Feedstock Supply	11.8%	Polymer Production	9.3%
Distribution	8.8%	Distribution	7.7%	End of Life	0.9%
Article Production	3.5%	Article Production	4.0%	Distribution	0.7%
End of Life	-3.8%	End of Life	-4.4%	Article Production	0.5%
<b>Particulate Matter</b>		<b>Acidification</b>		<b>Eutrophication - terrestrial</b>	
<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	60.6%	Feedstock Supply	63.7%	Feedstock Supply	68.9%
Polymer Production	33.9%	Polymer Production	30.8%	Polymer Production	24.9%
Distribution	5.0%	Distribution	5.3%	Distribution	5.7%
Article Production	2.6%	Article Production	2.0%	Article Production	1.2%
End of Life	-2.1%	End of Life	-1.8%	End of Life	-0.7%
<b>S5 – Starch/PBAT blend</b>					
<b>Climate Change</b>		<b>Resource Use - fossils</b>		<b>Particulate Matter</b>	
<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	70.9%	Polymer Production	90.1%	Polymer Production	64.8%
End of Life	11.9%	Distribution	8.9%	Feedstock Supply	19.1%
Distribution	10.5%	Article Production	4.6%	Distribution	14.3%
Article Production	4.3%	Feedstock Supply	1.3%	Article Production	7.5%
Feedstock Supply	2.4%	End of Life	-5.0%	End of Life	-5.7%

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S6 – PLA/PBAT blend					
Climate Change		Resource Use - fossils		Water Use	
Life cycle stage	Contrib.	Life cycle stage	Contrib.	Life cycle stage	Contrib.
Polymer Production	73.2%	Polymer Production	86.4%	Feedstock Supply	86.2%
Feedstock Supply	12.0%	Feedstock Supply	7.2%	Polymer Production	11.3%
Distribution	7.7%	Distribution	6.8%	Distribution	1.0%
End of Life	4.0%	Article Production	3.5%	End of Life	0.9%
Article Production	3.1%	End of Life	-3.9%	Article Production	0.6%
Particulate Matter		Acidification			
Life cycle stage	Contrib.	Life cycle stage	Contrib.		
Feedstock Supply	50.9%	Feedstock Supply	53.3%		
Polymer Production	42.5%	Polymer Production	40.1%		
Distribution	5.9%	Distribution	6.4%		
Article Production	3.1%	Article Production	2.4%		
End of Life	-2.5%	End of Life	-2.1%		
S7 - CO <sub>2</sub> -based PP					
Climate Change		Resource Use - fossils		Particulate Matter	
Life cycle stage	Contrib.	Life cycle stage	Contrib.	Life cycle stage	Contrib.
Feedstock Supply	51.0%	Feedstock Supply	74.5%	Feedstock Supply	45.2%
Polymer production	28.9%	Polymer Production	28.0%	Polymer Production	42.4%
Distribution	9.7%	Distribution	7.8%	Distribution	16.4%
End of Life	6.6%	Article Production	3.8%	Article Production	8.7%
Article Production	3.8%	End of Life	-14.1%	End of Life	-12.6%

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### 3 5.7.3 Interpretation of characterised results

4 In this section, the characterised potential impacts of the assessed scenarios (Figures  
5 5.8-5.10) are compared to evaluate the effects of using an alternative feedstock or  
6 material for a specific fossil-based polymer of reference used in food packaging film  
7 manufacturing. Therefore, the impacts of CO<sub>2</sub>-based PP packaging film are compared  
8 with those of fossil-based PP packaging film, while bio-based LDPE packaging film is  
9 compared with fossil-based LDPE packaging film. Due to their (barrier) properties, PLA,  
10 Starch-based and PLA-based packaging films can be considered to be mainly intended as  
11 potential replacements for (biaxially oriented) PP packaging film, which is thus  
12 considered as the main reference for comparison. The impacts of reference scenarios  
13 (fossil-based PP and LDPE packaging films) are also initially compared, to provide useful  
14 elements to potentially extend the comparative considerations provided below to any of  
15 such reference scenarios.

16 The comparison focuses on the sixteen, default impact categories considered in the  
17 Product Environmental Footprint (PEF) context and in this assessment. Potential impacts  
18 on Biodiversity are discussed separately, as being estimated via an endpoint indicator  
19 that is not recommended in the PEF framework, and partially relying on different impact  
20 assessment methods for the underlying midpoint-level impact categories. In the following

comparison, differences between scenario impacts lower than 10% were not considered significant, in light of the uncertainty associated with the developed life cycle models and the applied impact assessment models. Moreover, any comparative considerations related to Human Toxicity and Ecotoxicity impact categories need to be interpreted in light of the higher uncertainty of the underlying impact assessment models and results. The same applies, to a lower extent, also to Water Use and Land Use. Finally, it has to be reminded that the following comparative considerations refer to a functional unit requiring the achievement of a comparable tensile performance (and thus reflecting the tensile strength of each material as relevant technical property).

Focusing on the relative performance of food packaging films made of the two fossil-based reference materials (i.e. PP and LDPE), PP film shows a better performance in all impact categories. This is mainly because LDPE is used in a higher mass with respect to PP (i.e. 5.28 kg vs 2.70 kg per FU) to produce a film with a comparable tensile performance (tensile strength of LDPE is lower than PP for a similar film thickness, density of both materials being comparable). In most categories, the impact of LDPE film is 80% to 140% higher than PP film, due to an increased impact of all life cycle stages (and especially those of Feedstock Supply and/or Polymer Production). However, for Ionising Radiation a larger impact increase is observed (3.5 times) compared to PP film, due to the much higher contribution from the Polymer Production stage.

The use of bio-based LDPE derived from Brazilian sugarcane for packaging film manufacturing substantially increases the impact compared to the use of fossil-based LDPE in most impact categories. Exceptions are Resource Use – fossils (-114%), which is decreased thanks to the reduced use of fossil feedstock, Ozone Depletion (-96%), Water Use (-84%), and Ionising Radiation (-36%). The main responsible for the overall worsened performance is the increased feedstock supply impact, which (also) reflect the high specific sugarcane consumption for Bio-LDPE production (equalling 23.8 kg per kg of Bio-LDPE). The highest impact increases are observed for Particulate Matter (82 times) due to pre-harvest burning practice partially applied to sugarcane cultivation, Land Use (71 times) due to the increased land demand for biomass growing, and Eutrophication - marine (20 times) due to nutrient emissions from cultivation. A relevant increase is also observed for other (agricultural-related) impact categories, including Eutrophication Freshwater (9 times), Eutrophication Terrestrial (5 times), Acidification 4.5 times), as well as Human Toxicity – non-cancer 7 times) and Photochemical Ozone Formation (5 times). For the remaining categories, the increase is lower, but still significant, ranging between 17% (Resource Use – minerals and metals) and 90% (Human Toxicity – cancer).

PLA film shows significantly higher impacts compared to fossil-based PP film in the majority of the impact categories, despite the lower mass required per functional unit (i.e. 1.65 kg vs. 2.70 kg). Exceptions are the categories of Resource Use – fossils (where both alternatives are comparable) and Ozone Depletion, where PLA film is preferable (25% lower impact). The largest impact increases compared to PP film are observed for Land Use, Ecotoxicity – freshwater, Eutrophication (both freshwater and marine), and Water Use, where the impact of PLA is 7 to 30 times higher, mostly due to an increased impact from Feedstock Supply (i.e. maize cultivation in the US). For most of the other categories, the increase in impact is lower, but still significant (up to 5 times higher), while it is more limited for Climate Change (+34%), Photochemical Ozone Formation (+46%), and Human Toxicity – cancer (+71%).

A similar situation is also observed for Starch-based (TPS/PBAT) film, which in contrast to PLA film requires a higher mass per functional unit compared to PP to ensure a comparable tensile performance (i.e. 3.15 kg vs 2.70 kg). This generally leads to increased impacts throughout the whole life cycle of Starch-based film with respect to PP film, especially at the Polymer Production stage and (in some categories) at the one of Feedstock Supply<sup>73</sup>. However, in this case the observed increase is lower than PLA film in

<sup>73</sup> An increased impact from Feedstock Supply takes place in some of those categories more largely affected by agricultural production, including Eutrophication (terrestrial and marine), Land Use and Water Use. On the

many impact categories (and especially in those that are more largely affected by agricultural production activities)<sup>74</sup> due to the only partial bio-based content in the polymer blend (40%) compared to PLA (fully bio-based). In these categories, Starch-based film is preferable to PLA film (despite the latter has a lower mass), while in the remaining categories PLA film is instead preferable. The Polymer Production stage dominates the overall lifecycle impacts of Starch-based film in the majority of the impact categories (i.e. all except Human Toxicity – non-cancer, Ecotoxicity – freshwater, and Water Use). In many of these (9 out of 16), additives holds a large share of Polymer Production impacts, ranging between 21% and 80%. Considering this relevance, and the uncertainty associated with the modelling of additives (proportions and substances), a sensitivity analysis was performed to evaluate the effects of disregarding additives from the polymer composition (as detailed in Section 5.7.7.4).

PLA-based (PLA/PBAT) packaging film shows a worsened performance compared to fossil-based PP film across all impact categories, with most of them presenting an impact that is between 2.5 and 10 times higher than the mentioned reference alternative. However, in a few impact categories (dominated by Feedstock Supply) the observed impact increase is even larger, i.e. in Land Use (60 times), Ecotoxicity – freshwater (50 times), and Eutrophication – freshwater (30 times). This picture is mostly a consequence of the significantly increased mass of material required to fulfil the functional unit (i.e. 4.21 kg vs 2.70 kg), which is the highest among the three investigated biodegradable alternatives (i.e. PLA, starch-based and PLA-based film). Therefore, PLA-based film is the worst performing biodegradable alternative (excluding Ozone Depletion, where Starch-based film shows the worst performance).

Using CO<sub>2</sub> captured from coal-based power plants<sup>75</sup> and Hydrogen from the current mix of technologies as feedstock sources to produce PP is not proven to be beneficial for the environment (despite considering CO<sub>2</sub> as a waste for recycling from electricity production). CO<sub>2</sub>-based PP packaging film indeed shows higher impacts compared to fossil-based PP film across all impact categories except for Human Toxicity – cancer, where both alternatives are comparable. An impact increase between 23% and 128% is observed for the majority of the categories, due mostly to the increased impact from the stage of Feedstock Supply (which also include Hydrogen production) and/or Polymer Production. However, in some categories, a higher increase is observed. This is the case of Ionising Radiation 6 times, dominated by the impact from electricity used in Methanol synthesis), Ozone Depletion (5 times, dominated by chemicals and natural gas used in PP polymerisation), Land Use (5 times), and Resource Use – minerals and metals (3 times). However, compared to bio-based packaging films, CO<sub>2</sub>-based PP film shows a better performance in the majority of the impact categories. Only few of them are excluded, depending on the polymer (i.e. just one in the case of PLA/PBAT film, up to five in the case of PLA film).

Focusing on estimated potential impacts on biodiversity of the two fossil-based reference scenarios (Figure 5.15), LDPE packaging film shows a worse performance than PP film, mostly due to the large difference in the mass of the two polymers (and higher in the case of LDPE) rather than to real differences in supply chain drivers. Starch-based packaging film shows a disproportionately higher impact compared to fossil-based PP film and to all the other examined alternatives. This is due to the dramatically increased impact from (bio-based) Feedstock Supply, which mostly come from terrestrial ecotoxicity (endpoint) impacts from potato cultivation<sup>76</sup>, and, to a lower extent from wheat cultivation. The other biodegradable film alternatives (i.e. PLA and PLA/PBAT film) involve a higher impact compared to fossil-based PP film, especially when PLA/PBAT is

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other hand, the impact of Feedstock Supply is reduced in a number of categories that are normally less affected by agricultural production, including Climate Change, Human Toxicity – cancer, Ionising Radiation, Photochemical Ozone Formation, and Resource Use (both fossils and minerals and metals).

<sup>74</sup> These categories include Human Toxicity – cancer, Particulate Matter, Acidification, Eutrophication (terrestrial, freshwater and marine), Ecotoxicity – freshwater, Land Use and Water Use.

<sup>75</sup> Being the most abundant point emissions source at present.

<sup>76</sup> Dominated by insecticide emissions to soil (which are responsible for more than 90% of the total impact).



used as a film material (which in contrast to PLA requires a higher mass compared to PP film). This impact increase is a consequence of the increased impact of Feedstock Supply (dominated by land occupation impacts from maize cultivation) and, in the case of PLA/PBAT film, also of Polymer Production (where CO<sub>2</sub> emissions from PLA production play a major role, although the origin these cannot be investigated further). On the other, PLA packaging film has an impact comparable to fossil-based LDPE film (which has a substantially higher mass) and lower than CO<sub>2</sub>-based PP film. Finally, Bio-based LDPE packaging film shows a worse performance than its fossil-based counterpart, due to an increased impact from Feedstock Supply and the resulting land occupation for sugarcane cultivation. It has to be reminded, however, that direct potential biodiversity (ecosystem) impacts from oil leakage are not quantified for packaging films made of fossil-based polymers considered as a reference (although emissions from leakage per unit of oil supplied are reported to be quite small; see Section 4.4.1.1). This consideration also applies to the previous comparisons between bio-based and fossil-based alternatives.

As a last comment, it is noted how the Climate Change impact of bio-based food packaging films (i.e. Bio-LDPE, PLA, Starch-based and PLA-based films) is reduced if the contribution of non-released biogenic carbon after 100 years from landfilling and on-land application of composted organic material is taken into account. However, the reduction is significant only for bio-based LDPE film (-27%) due to the higher carbon content of this polymer (nearly 82%). In the case of PLA film (which has a carbon content of nearly 50%) the decrease is lower (10%), while for Starch/PBAT and PLA/PBAT films a reduction in the range of 4-5% is observed, due to the only partial bio-based content in such polymers.

#### **5.7.4 Interpretation of normalised and weighted results**

This section briefly compares the investigated scenarios based on the respective total impact scores, calculated by aggregating normalised and weighted impact assessment results across all impact categories (Table B.2.10 in Annex B.2). The comparison is carried out according to the same criteria outlined in Section 5.7.3 in terms of contrasted packaging film scenarios, and considering significant only impact differences larger than 10%. While affected by greater uncertainty (from the calculation and application of normalisation and weighting factors), total impact scores provides a more immediate and synthetic indication of the overall environmental performance of the analysed product scenarios. This facilitates the interpretation compared to looking at characterised results from several impact categories. However, such estimates are inevitably affected by values choices underlying the definition of weighting factors aimed at establishing an order of relevance for the different impact categories in a European decision context.

The comparison among total normalised and weighted scenario impacts reflects the overall picture of results emerged when focusing on characterised impact assessment results in single impact categories, and can be summarised in the following points:

- Fossil-based LDPE food packaging film shows a nearly doubled total impact score compared to fossil-based PP film, also reflecting the almost doubled mass of polymer required per functional unit to ensure a comparable (tensile) performance.
- Bio-based LDPE packaging film from Brazilian sugarcane shows a substantially worse performance compared to its fossil-based counterpart, with a total impact score that is four times higher than the latter.
- Biodegradable packaging films (PLA, Starch/PBAT, PLA/PBAT) show an increased overall impact compared to fossil-based PP film considered as a reference for these alternatives. The increase is especially relevant for PLA/PBAT film (almost four times), reflecting the higher mass required per functional unit compared to PP film. For PLA film (which instead requires a lower mass than PP film) and Starch/PBAT film, the increase is lower, but still significant (i.e. 72% and 99%, respectively).

- Using CO<sub>2</sub> captured from coal-based power plants and Hydrogen from the current mix of production technologies for PP packaging film manufacturing implies an overall worsened environmental performance compared to the use of fossil-based feedstock sources (+77% total impact score).

### 5.7.5 Effects of indirect land use change (iLUC)

When the contribution of GHG emissions from iLUC is taken into account, only small or moderate changes are identified in the Climate Change impact of bio-based alternatives (i.e. Bio-LDPE, PLA, Starch/PBAT and PLA/PBAT films) (Figure 5.8 and Table 5.17). A maximum increase of 12% is observed for bio-based LDPE film, while for Starch/PBAT film the increase is equal to only 0.4%, reflecting the only partial bio-based content of this polymer (i.e. 40%). For PLA and PLA/PBAT films, the increase is also small (i.e. 2.8% and 1.7%, respectively), despite they are totally (PLA) or predominantly (PLA/PBAT) of bio-based origin.

Overall, accounting for iLUC effects does not affect the relative performance of the assessed scenarios (i.e. their comparison is not affected). However, this outcome needs to be read also in light of the GHG emission factors applied for iLUC in this assessment, which appear to fall in the lower end of the range of values available in the recent literature (see Section 4.4.16.3 of the Method (Report I)). The use of an alternative iLUC model and of the resulting emission factors was explored as a sensitivity analysis (see Section 5.7.7.5).

### 5.7.6 Alternative End of Life options

This section discusses characterised scenario results calculated by individually applying each viable End of Life option as an exclusive (100%) End of Life scenario (as presented in Figures 5.12 to 5.14 and Tables B.2.11 and B.2.12 in Annex B.2). The main purpose is to evaluate how the performances of the single food packaging film scenarios are affected by the change in the applied End of Life option, rather than to mutually compare such scenarios for different End of Life options. However, the results should not be interpreted as a strict comparison among different End of Life alternatives for the studied article, since the evaluation applies a product perspective, and burdens/benefits of selected End of Life options (e.g. recycling) are partitioned between different product life cycles (which prevents from capturing the full implications of having a given waste stream routed to a specific End of Life option). In a waste management system perspective, where allocation is not needed, the total (system-wise) savings associated with the End of Life pathway “100% recycling” would likely be higher than what presented in this report, where a product perspective is applied.

None of the considered product End of Life scenarios (i.e. mechanical recycling, incineration, landfilling, composting and anaerobic digestion) can be identified as preferable across all the assessed impact categories and food packaging film scenarios. Moreover, in several categories, scenario impacts show no relevant changes when the different End of Life options are implemented individually (especially for bio-based packaging films), mostly due to the modest contribution of the End of Life stage to the overall scenario impacts. However, within this possibly limited range of variation, the following considerations can be made.

For packaging films made of fossil-based polymers (PP, LDPE), and CO<sub>2</sub>-based PP film, 100% mechanical recycling is the preferable scenario in six impact categories, including Climate Change, Human Toxicity – cancer, Human Toxicity – non-cancer, Eutrophication – freshwater, Ecotoxicity – freshwater and Water Use. For bio-based LDPE film, recycling is preferable only in three impact categories, i.e. Human Toxicity – cancer, Ecotoxicity – freshwater and Water Use. For biodegradable packaging films (PLA, Starch/PBAT and PLA/PBAT), recycling was not investigated, being not considered a viable End of Life option for these materials. In Ozone Depletion, mechanical recycling of all non-biodegradable packaging films shows the worst performance, as it also happens in Land

Use (except for fossil-based LDPE film, where landfilling is the worst scenario). In Ionising Radiation, recycling is almost comparable with landfilling, which is the least preferable scenario. For LDPE packaging film, this is also the case of Land Use. In the remaining categories, the recycling scenario has an intermediate performance between the incineration and landfilling scenarios.

For all packaging films alternatives, 100% incineration is the most favourable End of Life scenario in eight impact categories, including Particulate Matter, Photochemical Ozone Formation, Acidification, Eutrophication – terrestrial, Eutrophication – marine, Land Use, Resource Use – minerals and metals, and Resource Use – fossils. This is also the case of Ionising Radiation and Eutrophication – freshwater, if either PLA or CO<sub>2</sub>-based PP films are respectively excluded<sup>77</sup>. In the case of Ozone Depletion, the incineration scenario is still preferable for all packaging film alternatives except for those relying on Starch/PBAT and PLA/PBAT blends as film materials. For biodegradable packaging films (PLA, Starch/PBAT and PLA/PBAT), incineration is the preferable scenario also in Human Toxicity – cancer, Human Toxicity – non-cancer (including also Bio-LDPE film), and Ecotoxicity – freshwater (excluding PLA film). On the other hand, the incineration scenario shows the highest impact in terms of Water Use, Climate Change (limited to PP, LDPE, and CO<sub>2</sub>-based PP packaging films) and, in the case of PLA, also of Ecotoxicity – freshwater.

In many impact categories, 100% landfilling is the worst End of Life scenario for all or most packaging film alternatives, in line with the priority order outlined in the “Waste Hierarchy”, which sets disposal as the least preferable option (EC, 2008). Most relevant exceptions are Water Use, Ozone Depletion, and Climate Change, where other End of Life scenarios show the highest impact for all or many packaging film alternatives (as discussed above). Overall, the landfilling scenario is never preferable, although for PLA packaging film it is essentially comparable with most of the other scenarios in the Water Use impact category, and the same applies to Starch/PBAT packaging film in the Ecotoxicity – freshwater category. Moreover, if the contribution of non-released biogenic carbon after 100 years from landfilling is taken into account, 100% landfilling is the preferable scenario for bio-based packaging films (Bio-LDPE, PLA, Starch/PBAT and PLA/PBAT).

End of Life scenarios assessed for biodegradable packaging films (PLA, Starch/PBAT and PLA/PBAT) included also biological treatment options, i.e. composting and anaerobic digestion (both followed by on-land application of the residual organic material from such treatments). 100% composting is the preferable scenario only in a limited number of impact categories and only limited to some of the biodegradable packaging film alternatives. These include Ozone Depletion (Starch/PBAT and PLA/PBAT film), Human Toxicity – cancer (PLA film), Ecotoxicity – freshwater (Starch/PBAT film), and Water Use (PLA film). On the other hand, for all biodegradable films composting is the second worst End of Life scenario after landfilling in several categories, including Particulate Matter, Ionising Radiation, Photochemical Ozone Formation, Acidification, Eutrophication – terrestrial, Land Use, and Resource Use – fossils. Similar considerations apply to the anaerobic digestion scenario, which for some biodegradable film alternatives is preferable to the other scenarios only in terms of Ionising Radiation (PLA and PLA/PBAT film), Acidification (PLA and PLA/PBAT film), Eutrophication (PLA film), and Water Use (Starch/PBAT and PLA/PBAT film). At the same time, for all biodegradable films anaerobic digestion is the least preferable scenario in terms of Human Toxicity – cancer, Eutrophication – freshwater, Ecotoxicity – freshwater, and for Starch/PBAT film also of Climate Change, and Human Toxicity – non-cancer. However, as already discussed above, for bio-based packaging films differences among alternative End of Life scenarios are in most categories relatively small, so that any ranking needs to be interpreted within a limited range of impact variation.

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<sup>77</sup> For Ionising Radiation, anaerobic digestion of PLA is preferable (albeit comparable with incineration), while for Eutrophication – freshwater, mechanical recycling of CO<sub>2</sub>-based PP is preferable (but again comparable with incineration).

The Climate Change impact of the 100% composting and 100% anaerobic digestion scenarios was also calculated by accounting for the contribution of non-released biogenic carbon after 100 years from on-land application of the residual organic material obtained from such treatments. For graphical reasons, these results are not displayed in Figure 5.12. However, no significant changes are observed in the Climate Change impact of biodegradable packaging film alternatives (less than 1% decrease), due to the limited amount of residual, non-emitted carbon (equalling 1.5-3.2% of total carbon in the original polymer).

### **5.7.7 Interpretation of results related to macro- and micro-plastics generation**

This section discusses the results presented in Section 5.6.3 on the estimated potential generation of macro- and micro-plastics of the assessed food packaging films scenarios. The latter are compared according to the same criteria applied to the interpretation of the potential environmental impacts (Section 5.7.3), initially focusing on the results from the two-level approach applied as a base case. The results from the approaches applied as a sensitivity analysis are discussed separately.

Regarding the total loss of plastics (macro- and micro-plastics) and the resulting release to ocean and to the terrestrial environment (Figure 5.16), the two base-case approaches (PLP method and Expanded PLP method) provide similar results when comparing the different food packaging films scenarios. In both cases, the scenario considering the use of PLA as alternative feedstock sources for PP films show the best performance (-39%) in the total loss and release compared to the respective fossil-based counterpart. On the contrary, LDPE and Bio-LDPE show the highest values of plastic loss and release to the environment (+95% when compared to PP). This is mostly a consequence of the amount of polymer required per functional unit, which leads to a proportionally reduced or increased loss and release of plastics from both upstream life cycle stages (micro-plastics) and especially from product End of Life (macro-plastics). Note, however, that the applied methods rely on mass-based indicators (which are affected by differences in film mass) rather than quantifying the number of items (or plastic fragments/particles) lost or released to the environment. If this was the case, all scenarios would result in the same macro-plastics generation at End of Life when applying the PLP method, since films with a lower mass would not imply a reduction in the contribution of product littering and waste mismanagement (which are based on release rates that are not affected by the type of feedstock or material).

Comparing the two base-case approaches, the Expanded PLP method results in a higher plastic loss and release to ocean and the terrestrial environment compared to the original PLP method, equalling on average 15% for fossil-based scenarios (Figure 5.16). This is mainly due to an increased contribution of macro-plastics loss at End of Life, which also accounts for mismanagement of plastic waste exported for (sub-standard) recycling outside the EU (where the overall waste mismanagement share is increased by 2% compared to the assumption that all plastic waste is recycled in Europe). In the case of bio-based scenarios, this increase is not observed as films are composted instead of recycled at the End of Life. Conversely, the two additional micro-plastics sources considered in this alternative approach (road markings and marine coatings) does not significantly affect the results, due to the overall modest contribution of micro-plastics to the total release (in terms of mass).

Focusing on the relation between total loss and release (which depends on the parameters of the PLP method), the overall plastic loss is considered a release to the environment. In relative terms, release to ocean accounts for a lower share of the plastic loss (25%), while release to the terrestrial environment is larger (75%). As shown in Figure 5.18 for the PLP method, the total plastic release to both ocean and the terrestrial environment is dominated by the release of macro-plastics at the End of Life (more than 99%), while micro-plastics released throughout the upstream life cycle have only a minor role. Similar results are also observed for the Expanded PLP method (not shown), despite

two additional sources of micro-plastics are considered in the latter (i.e. road markings and marine coatings). In both cases, this is a consequence of the prevailing mass of macro-plastics, which is directly related to the mass of films required per functional unit (reference flow), by means of higher loss and release rates compared to micro-plastics. Therefore, the mass of macro-plastics is at least one order of magnitude higher than the mass of micro-plastics, which instead depends on the quantity of relevant lifecycle process by means of (much) lower loss and release coefficients (depending on the source).

Due to the prevailing role of macro-plastics, the comparison among the sole macro-plastics release to ocean (from product littering and waste mismanagement) estimated for food packaging films scenarios with both base-case approaches reflects the results obtained when looking at the total plastic release into such compartment (Figure 5.17). Comparative results are also in line with those of the sensitivity analysis based on marine litter rates, where PLA films still show the lowest release. However, the sensitivity analysis estimate smaller amount of plastic release to ocean due to the limited granularity of the database sourcing consumption intensity data. However, it must be reminded that all the approaches rely on mass-based indicators, and the resulting trends are thus determined by the mass of material required to fulfil the functional unit. This implies that scenarios relying on lighter films provide a lower contribution to macro-plastics generation (and vice versa), which would not be the case if an indicator based on the number of released items was considered. Note, however, that such estimates only provides information at the level of product category (i.e. films) without accounting for specific product properties (e.g. type of polymer) and for the specific supply-chain configuration (e.g. in terms of type and location of waste treatment operations). For all scenarios, emissions from product littering represent around a third of the total macro-plastics release, between 31% and 36%, highlighting the important role of waste mismanagement.

With regard to the origin of released plastics (in terms of life cycle stage and source), macro-plastics entirely originate from the End of Life stage (Figure 5.19), as they are only derived from product littering or mismanagement of the product as waste. On micro-plastics release, the relative contribution of the different sources (e.g. loss of plastic pellets and tire abrasion) depends on the characteristics of the supply-chain (e.g. feedstock origin, which affect the contribution from transport across the life cycle), as well as on the mass of polymer required per functional unit (which affect, to different extents, all the considered sources). However, within the limited role of micro-plastics in the overall release to the environment, the most relevant contribution is generally provided by pellet losses from the Polymer and Article Production stages, while transport-related sources have a more restricted role. An exception is PLA films, where transport emissions are more important due to increased transport throughout the supply chain.

In conclusion, due to the dominant role of macro-plastics loss and release at the End of Life of packaging food films, a best overall performance is associated to the scenario requiring a lower amount of plastic to fulfil the functional unit (i.e. relying on lighter films; mass-driven results). In this context, the use of PLA films is the scenario with the lowest plastic loss and release to both ocean and the terrestrial environment, while regardless of the feedstock LDPE films involve a higher plastic loss and release than PP films. The same considerations apply to the sole release of macro-plastics to ocean at End of Life, although in this case the use of an indicator based on the number of items (i.e. films) released would provide the same result regardless of the scenario when applying the PLP method or the sensitivity estimates based on marine litter rates. Films with different masses would indeed equally contribute to littering and waste mismanagement at End of Life. Conversely, an items-based indicator would still show different results for fossil-based films in the Expanded PLP method, with a higher release of macro-plastics due to mismanagement of waste exported for recycling in non-EU countries. From a methodological perspective, the different approaches provide similar results when comparing alternative scenarios.

## 5.7.8 Sensitivity analysis

A sensitivity analysis has been performed on a number of relevant parameters or assumptions, to evaluate the effects of their variation on the potential impacts of the affected scenario(s), and on the comparative LCA results. The following aspects have been specifically considered:

1. Alternative reference flow;
2. "Slash and burn" rate for sugarcane used as a feedstock for bio-based LDPE food packaging film;
3. Feedstock source for bio-based LDPE food packaging film (origin and type) and PLA film (origin and resulting PLA production country);
4. Starch content and use of additives in Starch-based food packaging film;
5. CO<sub>2</sub> source and Hydrogen supply technology for CO<sub>2</sub>-based PP food packaging film;
6. Modelling of CO<sub>2</sub> as a feedstock for polymer production;
7. iLUC model and factors.

The following sections present the outcome of the sensitivity analysis for each of the aspects above. The results are reported by comparing recalculated impacts of the affected scenario(s) with those of the respective base case assessment, of the corresponding reference scenario(s), and, where relevant, of the alternative feedstock scenarios evaluated for the same reference material. In case of changes in parameters contemporarily affecting the impacts of more scenarios, they are all taken into account in the presentation of the results, along with the respective reference scenarios.

### 5.7.8.1 Alternative reference flow

As a base case, scenario impacts were calculated based on polymer demands (reference flows) estimated assuming that the prioritised (or limiting) performance requirement (out of those expressed in the functional unit) is the need for the film of not breaking during transport and handling under a minimum specified tensile load (i.e. of ensuring a minimum tensile strength, as better discussed in Section 5.2). However, prioritising alternative performance requirements may lead to different reference flows (and thus results) since the compared materials perform differently towards the different technical properties affecting their ability to fulfil such requirements. Therefore, reference flows and the corresponding scenario impacts were recalculated considering the need of ensuring a comparable shelf life (and hence oxygen barrier/permeability)<sup>78</sup> as the priority performance requirement to be fulfilled, in order to evaluate the robustness of the results obtained as a base case.

The reference flow was calculated by means of Material Substitution factors (MSFs) and Material Indices (MIs; Ashby, 1999), following the approach already applied as a base case and described in Section 5.2. No specific expressions of MIs are reported by Ashby (1999) to score alternative materials against their gas permeability performances (due to the focus of the Author on mechanical performances). Therefore, a new expression was defined following the same underlying philosophy, but replacing specific mechanical properties with relevant material properties towards oxygen permeability (i.e. the Oxygen Transmission Rate – OTR). Considering that both a reduced OTR and density concur to either an improved material performance, or to a reduced material mass, MIs were calculated as the reciprocal of the product between these two material properties (i.e.  $MI = (OTR \cdot \rho)^{-1}$ ). The main calculations carried out to determine the reference flow for each scenario are summarised in Table 5.20, along with the values assumed for relevant material properties. As in base case calculations, the average oxygen

<sup>78</sup> As discussed in Section 5.2, shelf life was assumed to be exclusively determined by the oxygen permeability of the different packaging film alternatives.

permeability of non-oriented PP film with a typical median thickness of 30  $\mu\text{m}$  was considered as the reference performance to be achieved (corresponding to an OTR equal to 3040  $\text{cm}^3/\text{m}^2\text{-day}$ )<sup>79</sup>.

**Table 5.20.** Calculation of the reference flow for food packaging film LCA scenarios when oxygen barrier (determining shelf life) is the prioritised performance requirement to be fulfilled (the calculation is based on Material Substitution Factors).

Polymer	Density - $\rho$ ( $\text{kg}/\text{m}^3$ )	OTR <sup>(1)</sup> ( $\text{cm}^3/\text{m}^2/\text{d}$ )	MI ( $\text{OTR}/\rho$ ) <sup>-1</sup>	MSF ( $\text{m}/\text{m}_{\text{ref}}$ )	Reference flow ( $\text{kg}/\text{FU}$ )	Thick-ness ( $\mu\text{m}$ ) <sup>(2)</sup>
PP (all types of feedstock)	900 <sup>(3)</sup>	3040 <sup>(4)</sup>	0.296	1	2.70 <sup>(5)</sup>	30
LDPE (all types of feedstock)	928 <sup>(3)</sup>	7400 <sup>(6)</sup>	0.125	2.36	6.40	69
PLA (bi-axially oriented)	1,240 <sup>(7)</sup>	833 <sup>(7)</sup>	1.49	0.199	0.537	4.3
Starch/PBAT blend	1,220 <sup>(8)</sup>	2350 <sup>(8)</sup>	0.519	0.570	1.54	13
PLA/PBAT blend	1,290 <sup>(9)</sup>	2300 <sup>(10)</sup>	0.561	0.528	1.43	11

<sup>(1)</sup> OTR: Oxygen Transmission Rate

<sup>(2)</sup> The thickness of PP film (reference material) was assumed (and used to calculate the reference flow, see note 5). For films made of the other materials, thickness was backward calculated for checking purposes, based on the estimated reference flow and the assumed film extension (100  $\text{m}^2$ ). The estimated thickness values are realistic, although the quite low value obtained for PLA film falls outside the typical range of thicknesses offered on the market for film made of this material (20-100  $\mu\text{m}$ ).

<sup>(3)</sup> Average of the range (extreme values) reported in the review by Mangaraj et al. (2009).

<sup>(4)</sup> Value of OTR reported in Robertson (2011) for a film with a thickness of 25  $\mu\text{m}$  (which is very close to the value of thickness assumed as a reference for calculation purposes).

<sup>(5)</sup> For PP film (reference material), the reference flow is calculated assuming a film thickness of 30  $\mu\text{m}$ , representing the typical median value of the thickness range offered for some (BO)-PP films available on the market.

<sup>(6)</sup> Value of OTR reported in Robertson (2011) for a film with a thickness of 25  $\mu\text{m}$ .

<sup>(7)</sup> Average of the values from the technical specifications of two BO-PLA packaging films available on the market. OTR values considered to calculate the reported average value refer to films with a thickness of 30  $\mu\text{m}$ .

<sup>(8)</sup> Based on the technical specifications of a starch-based packaging film. OTR is calculated as the average of two values referring to films with a thickness of 35 and 40  $\mu\text{m}$ , respectively.

<sup>(9)</sup> Estimated based on linear regression on density values of a 100% PLA film (1240  $\text{kg}/\text{m}^3$ ) and of a PLA/PBAT film with a 34% PLA content (1340  $\text{kg}/\text{m}^3$ ).

<sup>(10)</sup> Based on the experimental results from Briassoulis and Giannoulis (2018), which refer to a 30  $\mu\text{m}$  PLA/PBAT film.

The results of the sensitivity analysis are shown in Figure 5.16, where scenario impacts are reported as a function of the impacts of fossil-based PP food packaging film, which is set as a 100% baseline since the reference flow and the impacts of this scenario do not change compared to the base case assessment. The impacts of CO<sub>2</sub>-based food packaging film are not presented, these being also not affected by this sensitivity analysis (nor is their comparison with the impacts of conventional, fossil-based PP film). However, recalculated impact values for each of the analysed packaging film scenarios are available in Annex B.2.

Similarly to the base case assessment, fossil-based LDPE film still show a worse performance compared to fossil-based PP film across all impact categories. This is due to the even higher amount of polymer now required to ensure a comparable oxygen barrier performance, which is a consequence of the poorer barrier properties of LDPE. On the

<sup>79</sup> This value actually refer to a film with a thickness of 25  $\mu\text{m}$ , which is considered sufficiently comparable to that assumed as a reference for calculation purposes (i.e.  $\mu\text{m}$ ).

other hand, the comparison between fossil-based and bio-based LDPE films is not affected by the sensitivity analysis, since the reference flow of both alternatives is subject to the same variation.

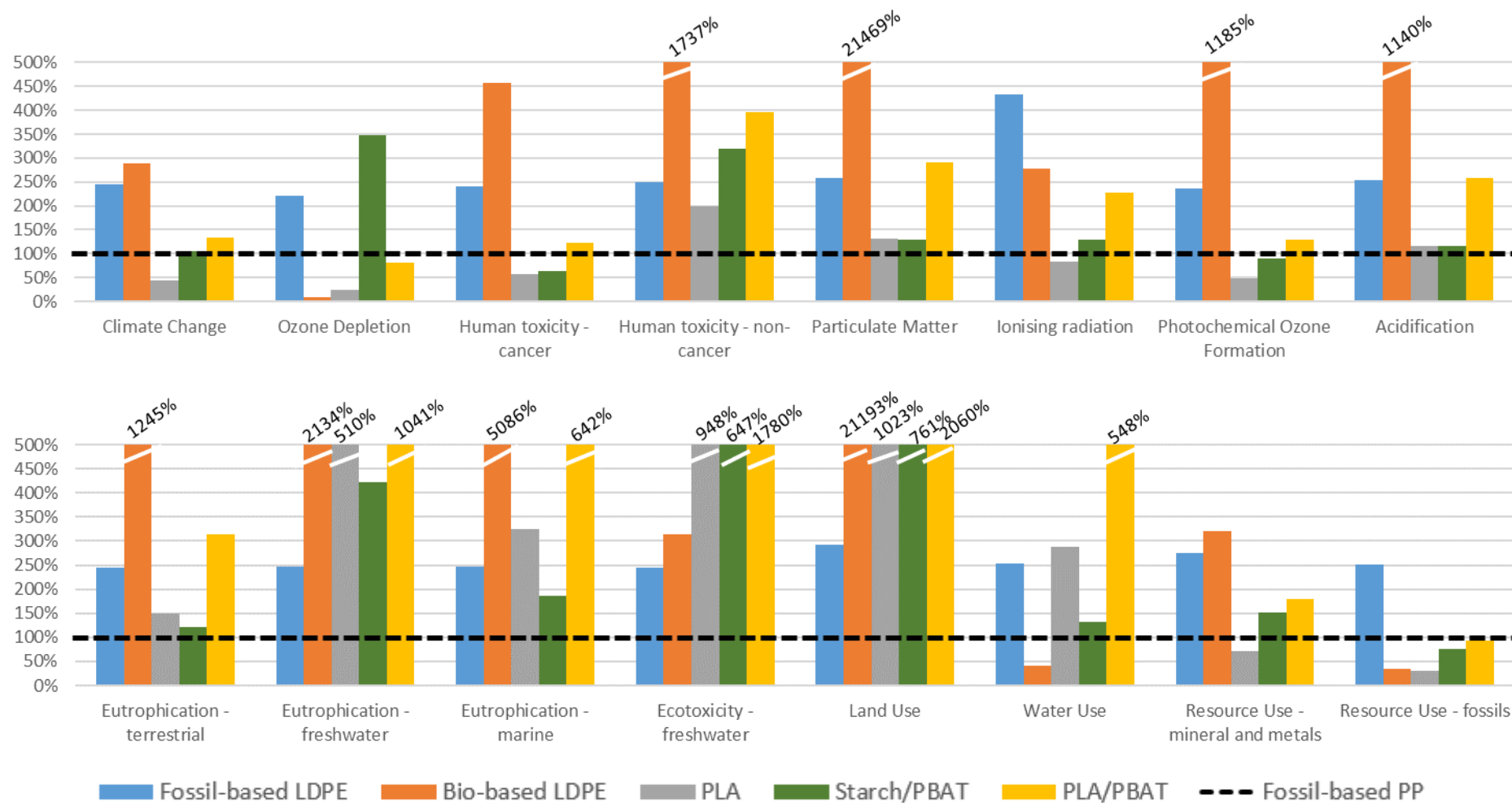
For biodegradable food packaging films (PLA, Starch/PBAT, and PLA/PBAT) the amount of polymer required per functional unit is now lower (or further lowered) compared to non-oriented PP packaging film, thanks to their better oxygen barrier performance (which compensate for the generally higher material density). This is especially the case of PLA packaging film, whose amount per functional unit is five times lower than PP packaging film, while the total mass of material estimated for starch-based and PLA-based packaging film is nearly half of that required for PP film.

In this framework, PLA packaging film now outperforms PP packaging film in several impact categories (Climate Change, Human Toxicity – cancer, Ozone Depletion, Ionising Radiation, Photochemical Ozone Formation, Resource Use – minerals and metals, and Resource Use – fossils), with an impact reduction between 18% (Ionising Radiation) and 76% (Ozone Depletion). However, in the nine remaining categories, PLA film still shows a (substantially) higher impact, despite the significantly lower mass.

Similarly, starch-based packaging film still shows a worse performance compared to fossil-based PP film in the majority of the impact categories, and the same applies to PLA/PBAT film. Starch-based packaging film is indeed preferable to PP film only in three impact categories (Human Toxicity – cancer, Photochemical Ozone Formation, and Resource Use – fossils) while being comparable to the latter in the case of Climate Change. PLA/PBAT packaging film instead outperforms fossil-based PP film only in the Ozone Depletion impact category, while showing a comparable performance in Resource Use – fossils.



## Comparative LCA of Alternative Feedstock For Plastics Production – DRAFT FOR STAKEHOLDER CONSULTATION –Part II



1     **Figure 5.16.** Results of the sensitivity analysis on the prioritised performance requirement to calculate the reference flow of food packaging film LCA  
2     scenarios (i.e. ensuring a comparable shelf life as determined by oxygen barrier). Scenario impacts are expressed as a percentage of the impacts of  
3     fossil-based PP food packaging film, which is not affected by the explored change and is set as a baseline. Note that in some impact categories a part of  
4     the results is out of scale and is curtailed.

### 5.7.8.2 “Slash and burn” rate for sugarcane and feedstock sourcing for Bio-LDPE

This sensitivity analysis explores the effects of completely removing the “slash and burn” practice in the cultivation of sugarcane used as a feedstock for bio-based LDPE production. 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 “slush and burn” rate was decreased from 45% (assumed as a base case) to 0%.

In addition, this analysis evaluates the effects of replacing Brazilian sugarcane with biomass feedstock sourced in Europe for bio-based LDPE production. The current EU-average mix of bioethanol crops was considered, which was estimated to include maize (45%), wheat (34%) and sugar beet (21%), based on bioethanol volumes obtained from such feedstock sources (ePURE, 2019). These shares applies identically also at the polymer level, since the conversion efficiency of bioethanol to Bio-LDPE is the same regardless of the feedstock used for Ethanol production. The feedstock substitution was modelled by means of vertically aggregated inventories available in the GaBi database, referring to Bio-LDPE production from the three types of alternative feedstock considered in this sensitivity analysis. Since the original dataset for maize-based Bio-LDPE refers to maize grown in the US, it was adjusted by subtracting the burdens associated with maize cultivation in US, to subsequently add those from maize cultivation in Europe. For this purpose, a specific maize consumption of 4.69 kg/kg Bio-LDPE was considered<sup>80</sup>, since the actual amount applied in the dataset is not disclosed. To represent the burdens of both US and European maize cultivation, two datasets from the GaBi database were applied, in the attempt to keep consistency with the originally applied dataset (which comes from the same database, despite being unknown). Transoceanic transport of US maize by ship was also initially subtracted, based on an estimated distance of 6000 km (the original assumed distance is also unknown). However, since the results were not significantly different compared to keeping transport included, this change was not ultimately implemented to avoid introducing an additional source of uncertainty.

Figure 5.17 shows the results of both changes applied to bio-based LDPE food packaging film. Scenario impacts are expressed as a percentage of those of the base case of the affected food packaging film scenario (i.e. bio-based LDPE packaging film with Brazilian sugarcane as a feedstock and a “slash and burn” rate of 45%), which is set as a 100% baseline.

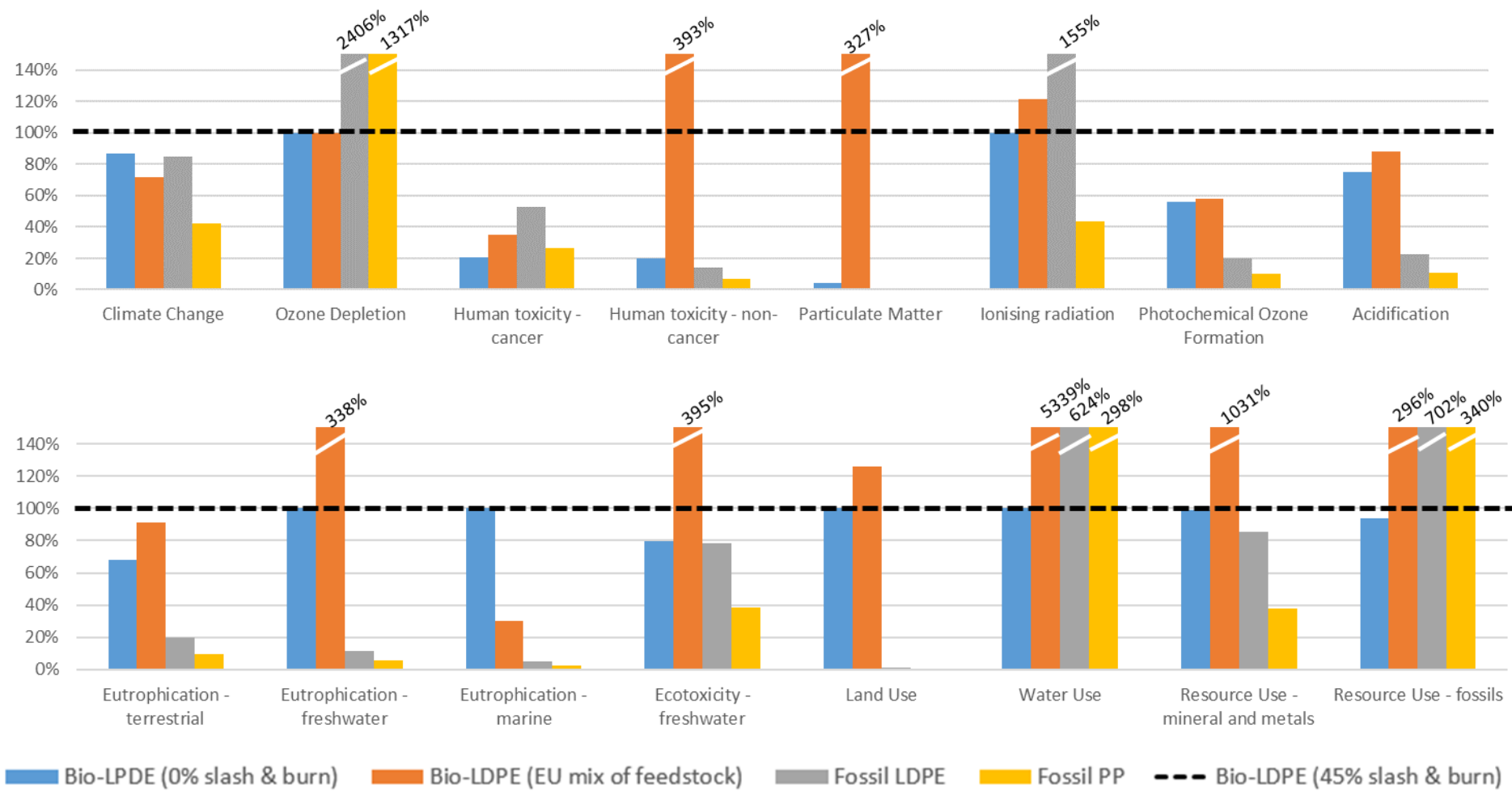
Completely phasing out the sugarcane pre-harvest burning practice decreases the impact of bio-based LDPE packaging film in eight impact categories, while no or irrelevant changes takes place in the remaining ones. The largest reductions are observed in Particulate Matter (-96%), Human Toxicity – non-cancer (-80%), Human Toxicity – cancer (-79%), and Photochemical Ozone Formation (-44%). In spite of these reductions, no substantial changes occur in the comparison with the reference scenario (i.e. fossil-based LDPE film), which is still preferable than, or comparable to bio-based LDPE film in the majority of the impact categories. However, compared to the base case (considering a 45% “slash and burn” rate), bio-based LDPE film is now preferable to fossil-based LDPE film in five categories instead of four (i.e. Ozone Depletion, Resource Use – fossils, Human Toxicity – cancer, Ionising Radiation and Water Use).

Replacing Brazilian sugarcane with the European mix of bioethanol crops increases the impacts of bio-based LDPE packaging film in eight impact categories, with most of them showing an even important increase (in the range of 227%-295%). In six categories the impact is decreased, while for the remaining ones the observed variations are irrelevant (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 explanation may

<sup>80</sup> The estimate is based on a specific consumption of starch for Bio-LDPE production equal to 4.95 kg starch/kg Bio-LDPE (IfBB Hannover, 2018), and a maize requirement of 0.948 kg maize grain/kg starch (according to maize starch-related datasets in the Agrifootprint database).

1 be the higher net energy demand of the processes required for the conversion of starch-  
2 based crops (maize, wheat) to bioethanol (compared to sugarcane), and the higher share  
3 of such crops in the applied mix compared to sugar beet (which requires a less energy  
4 intensive processing). This aspect is also reflected in available life cycle inventory  
5 datasets for bioethanol production (e.g. from the *ecoinvent* database), which show a  
6 larger impact of maize-based Ethanol compared to sugarcane in many impact categories.  
7 On the other hand, Ethanol from sugar beet provides lower impacts in many categories  
8 compared to sugarcane, but its share in the mix is lower (21%). Differences in cultivation  
9 practices applied in Europe to the alternative bioethanol feedstock sources considered in  
10 this analysis may also play a role. Overall, bio-based LDPE packaging film still shows a  
11 worsened performance compared to fossil-based LDPE film in the majority of the impact  
12 categories. Exceptions are Climate Change, Ozone Depletion, Human Toxicity – cancer,  
13 Ionising Radiation, and Resource Use - fossils.

14



**Figure 5.17.** Results of the sensitivity analysis on the sugarcane “slash and burn” rate and on feedstock sourcing for bio-based LDPE food packaging film. Scenario impacts are expressed as a percentage of the impacts of the base case of the affected scenario (i.e. bio-based LDPE food packaging film from Brazilian sugarcane with a 45% “slash and burn” rate), which is set as a baseline. Note that in some impact categories a part of the results is out of scale and is curtailed.

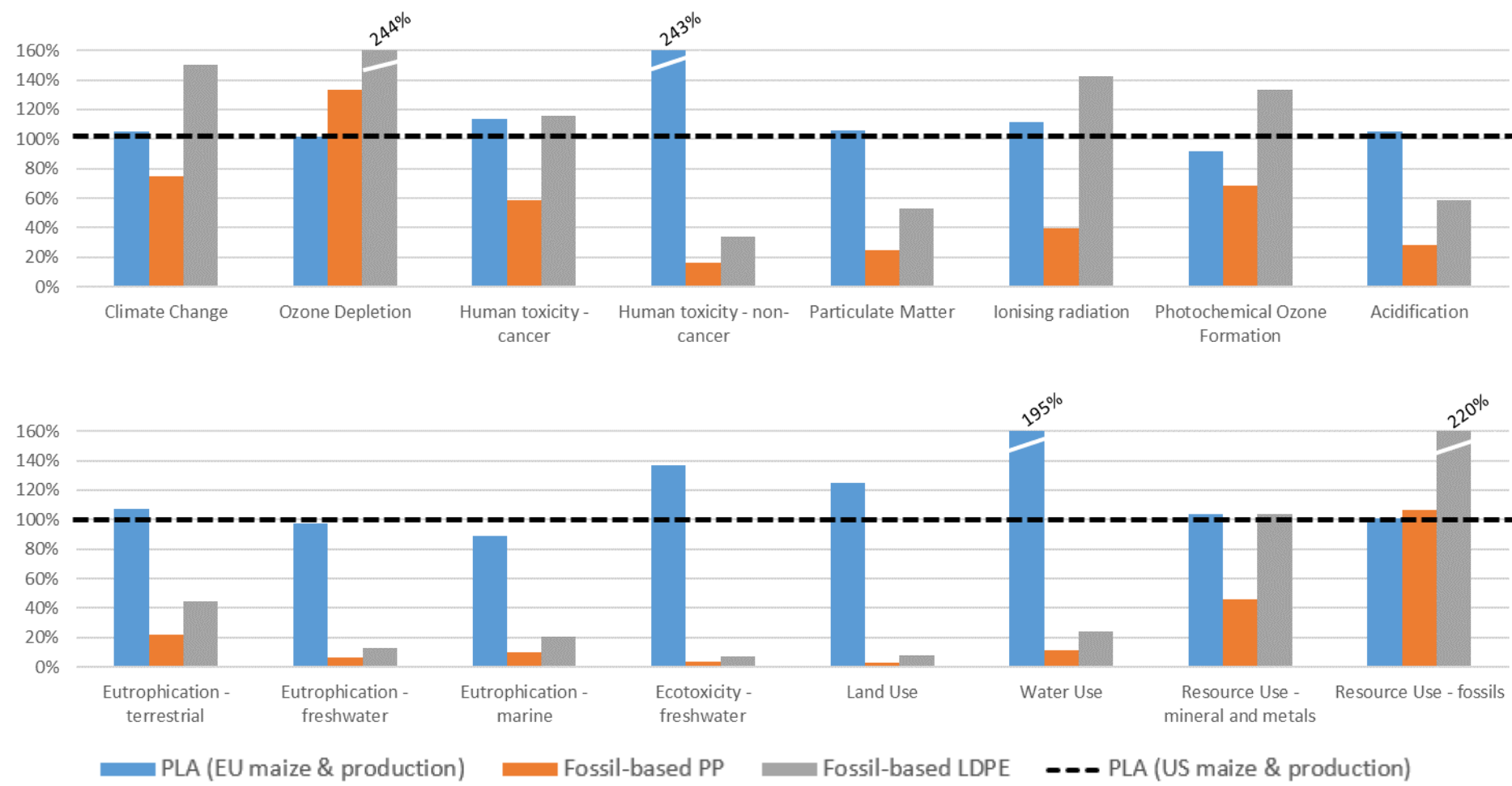
### **5.7.8.3 Feedstock origin and production country of PLA**

This sensitivity analysis evaluates the effects of relying on maize grown in Europe rather than in the US for the production of PLA used in both “neat” PLA and PLA/PBAT food packaging films. 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 previously applied EF dataset for maize cultivation in the US was replaced with the respective European equivalent, while keeping the same 100 km distance for subsequent lorry transport of maize to downstream processing. The diesel input to such transport was also adjusted to reflect the average diesel supply mix for Europe, rather than for US. The dataset applied to model PLA production from maize already reflected background European conditions, so that no adjustments were performed.

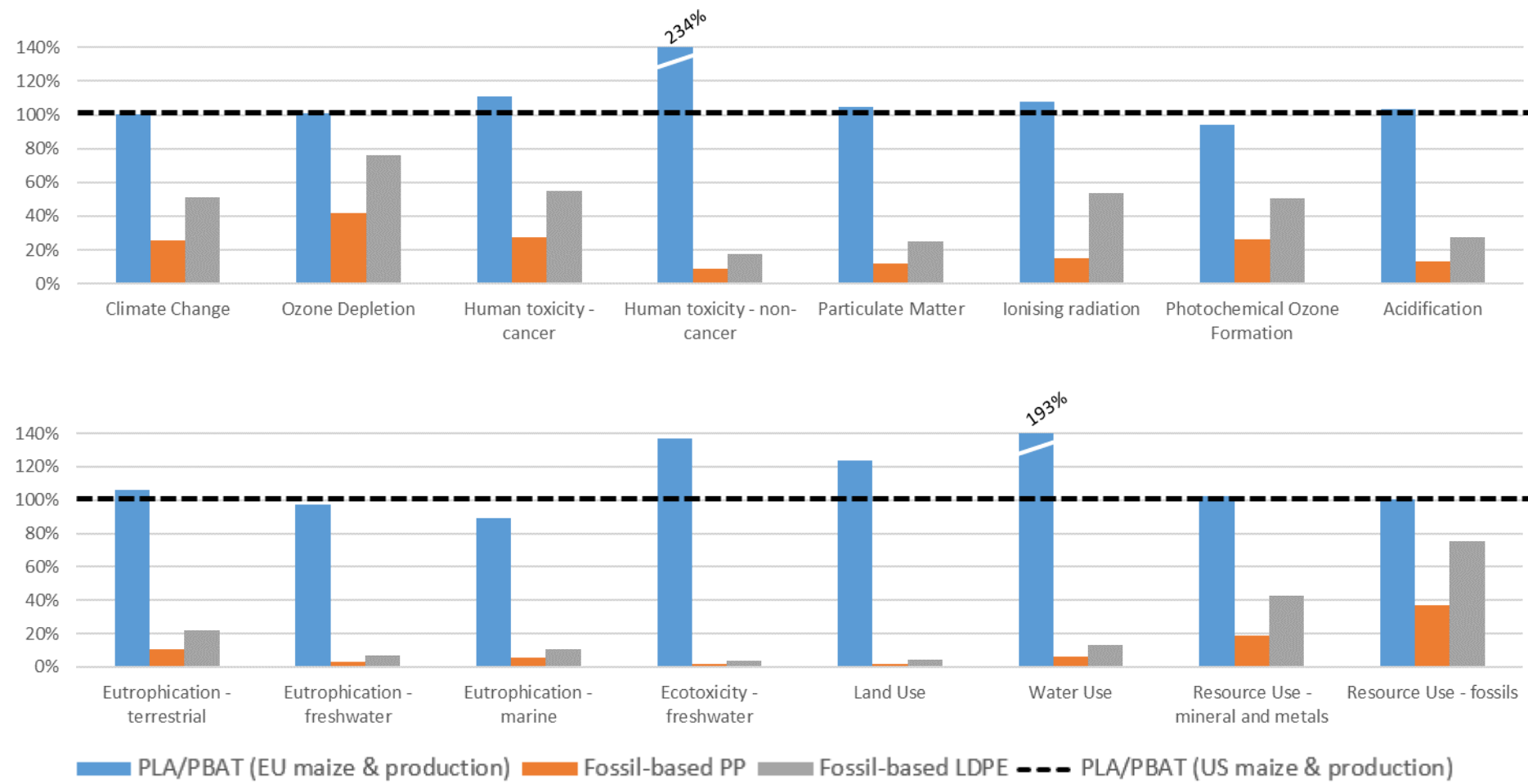
The results of the analysis are presented in Figures 5.18 (for PLA packaging film) and 5.19 (for PLA/PBAT packaging film), setting the base case of the two food packaging film scenarios (considering maize cultivation and PLA production in the US) as a 100% baseline.

Producing PLA in Europe from maize grown in the same region increases the impacts of PLA food packaging film in six impact categories, while only in one category the impact is decreased (Eutrophication – marine). The highest increase is observed for Human Toxicity – non-cancer (+143%) and the lowest one for Ionising Radiation (+12%). In the remaining categories, no significant impact variations are observed with the applied change in feedstock and polymer origin. The comparison with fossil-based PP packaging film, is also similar to the base case assessment, with PLA film outperforming PP film in only one impact category (Ozone Depletion) and showing a comparable performance in another category (Resource Use – fossils). In all the remaining categories, fossil-based PP packaging film still has a better performance.

Similar effects can be observed also for PLA/PBAT packaging film when the considered change in feedstock and polymer origin is applied to PLA used in the blend. In this case, the total lifecycle impacts of this packaging film alternative are increased in five categories, decreased in one, while being comparable in all the remaining categories. Moreover, PLA/PBAT packaging film relying on European PLA still has a worse performance than fossil-based PP film across all the assessed impact categories.



**Figure 5.18.** Results of the sensitivity analysis on feedstock sourcing for PLA food packaging film. Scenario impacts are expressed as a percentage of the impacts of the base case of the affected scenario (i.e. PLA food packaging film from PLA produced in the US from maize grown in the same country), which is set as a baseline. Note that in some impact categories a part of the results is out of scale and is curtailed.



**Figure 5.19.** Results of the sensitivity analysis on feedstock sourcing for PLA/PBAT food packaging film. Scenario impacts are expressed as a percentage of the impacts of the base case of the affected scenario (i.e. PLA/PBAT food packaging film from PLA produced in the US from maize grown in the same country), which is set as a baseline. Note that in some impact categories a part of the results is out of scale and is curtailed.



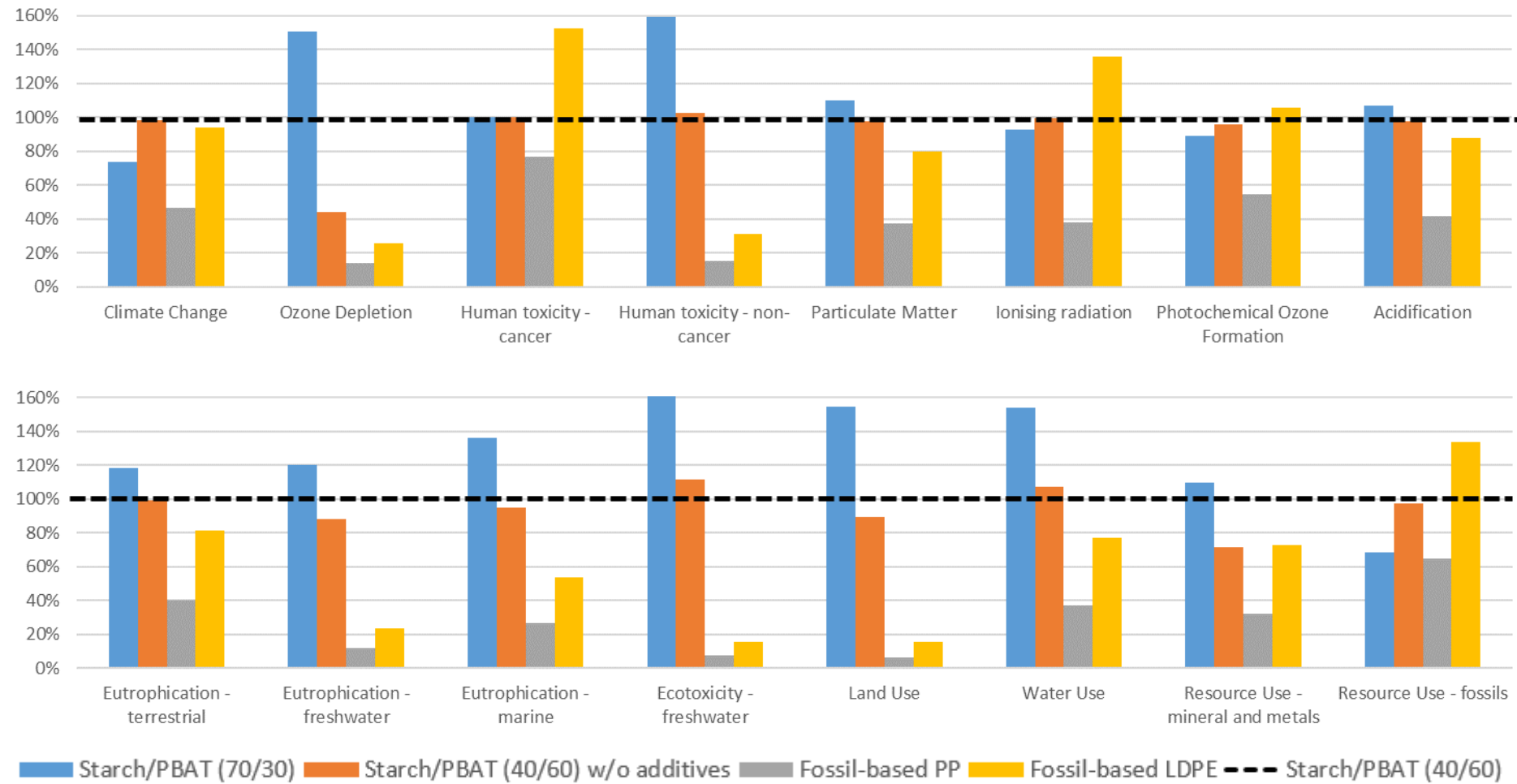
#### **5.7.8.4 Starch content and use of additives in the starch/PBAT blend**

As a base case, a 40% starch content was considered for Starch-based food packaging film, according to stakeholder's inputs reporting that "renewability" of starch-based polymers used in packaging applications can currently reach 40% (Section 5.1). However, recent reconstructions of mass flows in selected bio-polymer processing routes includes also (fictional) examples of starch-based polymers considering higher starch contents (i.e. 50% and 70%; IfBB, 2018). Therefore, this sensitivity analysis evaluates the effects of increasing the starch content to the highest value reported (70%), although this possibility is considered unlikely at present and may not be reflected in real products on the market. In addition, the effects of excluding additives (plasticisers) from the composition of the starch (TPS) copolymer were also explored, considering the uncertainty associated with their modelling (type of substances and related proportions assumed), and their non-negligible contribution to overall polymer production impacts in several impact categories. For the purpose of simplicity, the exclusion was limited to the production stage (by totally replacing additives with starch), while for End of Life modelling the composition was not changed (and could not be changed for those options modelled through aggregated datasets such as incineration). This choice is not expected to have significant effects on the results, since no hazardous substances are included as additives, but only (organic) substances presenting shares of C, H and O in the respective composition which are comparable to those in starch.

The results of the analysis are presented in Figure 5.20, setting the base case of the Starch-based food packaging film scenario (40% starch content and production of starch additives accounted) as a 100% baseline.

Increasing the starch content from 40% to 70% decreases the impact of Starch-based packaging film only in three impact categories, including Resource Use – fossils (-31.5%), Climate Change (-26%), and Photochemical Ozone Formation (-11%). For most of the remaining categories, the impact is increased (from 18% to 67%), or is not significantly affected (in five categories). Compared to fossil-based PP packaging film, Starch-based film with a 70% starch content still shows a worse performance in all impact categories, except for Resource Use – fossils, where both alternatives are comparable.

When additive production is excluded from the life cycle of Starch-based food packaging film (and totally replaced with starch production), no relevant changes are observed in the majority of the impact categories compared to the base case assessment. Few categories show an impact reduction (Ozone Depletion, Resource Use – minerals and metals, Eutrophication – freshwater and Land Use), while Ecotoxicity – freshwater shows a 12% increase in the impact. Also in this case, the comparison with fossil-based PP packaging film is still in favour of the latter, with Starch-based food packaging film presenting a worse performance across all impact categories.



**Figure 5.20.** Results of the sensitivity analysis on the starch content and use of additives in Starch/PBAT food packaging film. Scenario impacts are expressed as a percentage of the impacts of the base case of the affected scenario (i.e. Starch/PBAT food packaging film with a 40% starch content and including additives), which is set as a baseline.

#### 5.7.8.5 CO<sub>2</sub> source and Hydrogen supply technology for CO<sub>2</sub>-based PP

This sensitivity analysis explores the effects of options that may reduce the potential impacts associated with the production of CO<sub>2</sub>-based polyolefines, and especially CO<sub>2</sub>-based PP used, in this case study, for food packaging film manufacturing. First of all, the use of an alternative, more concentrated (but less abundant) stationary CO<sub>2</sub> source was assessed, considering CO<sub>2</sub> derived from Ammonia synthesis plants rather than coal-based power plants assumed as a base case. Moreover, the use of Hydrogen entirely produced from water electrolysis based on renewable energy was explored as an additional improvement option. Indeed, the potential larger availability of “low-carbon” Hydrogen in the future (provided that energy decarbonisation is achieved to a large degree) is one of the main driver for investigating CO<sub>2</sub>-based pathways. Note, however, that at present this scenario would reflect only (very) specific situations, which could equally apply to any other industrial production process (and not only Hydrogen production).

CO<sub>2</sub> supply from Ammonia production plants was modelled considering the activities of purification, liquefaction (compression) and transport to downstream users. Extraction (capture) from the gaseous mixture of CO<sub>2</sub> and Hydrogen generated in the Ammonia production process was excluded, since CO<sub>2</sub> removal from such a stream is a required step of the process itself (to isolate Hydrogen), which is conducted even in the absence of any further utilisation process. Hence, a 100% CO<sub>2</sub> stream is the actual waste output from Ammonia production. Conversely, in the case of sourcing from coal-based power plants, CO<sub>2</sub> is diluted in the generated flue gases, and capture/extraction is purposefully carried out to allow downstream utilisation, thus representing the first processing step of the “waste CO<sub>2</sub>” recycling chain. The burdens of the purification and liquefaction processes were modelled building upon the same adjusted *ecoinvent* dataset applied in the base case assessment for CO<sub>2</sub> sourced from coal-based power plants (Section 5.4.1.3). However, the electricity and heat consumption was further adjusted to cover only the two mentioned process steps<sup>81</sup>, while also removing the Monoethanolamine input (used only for capture) and airborne fugitive emissions of this substance.

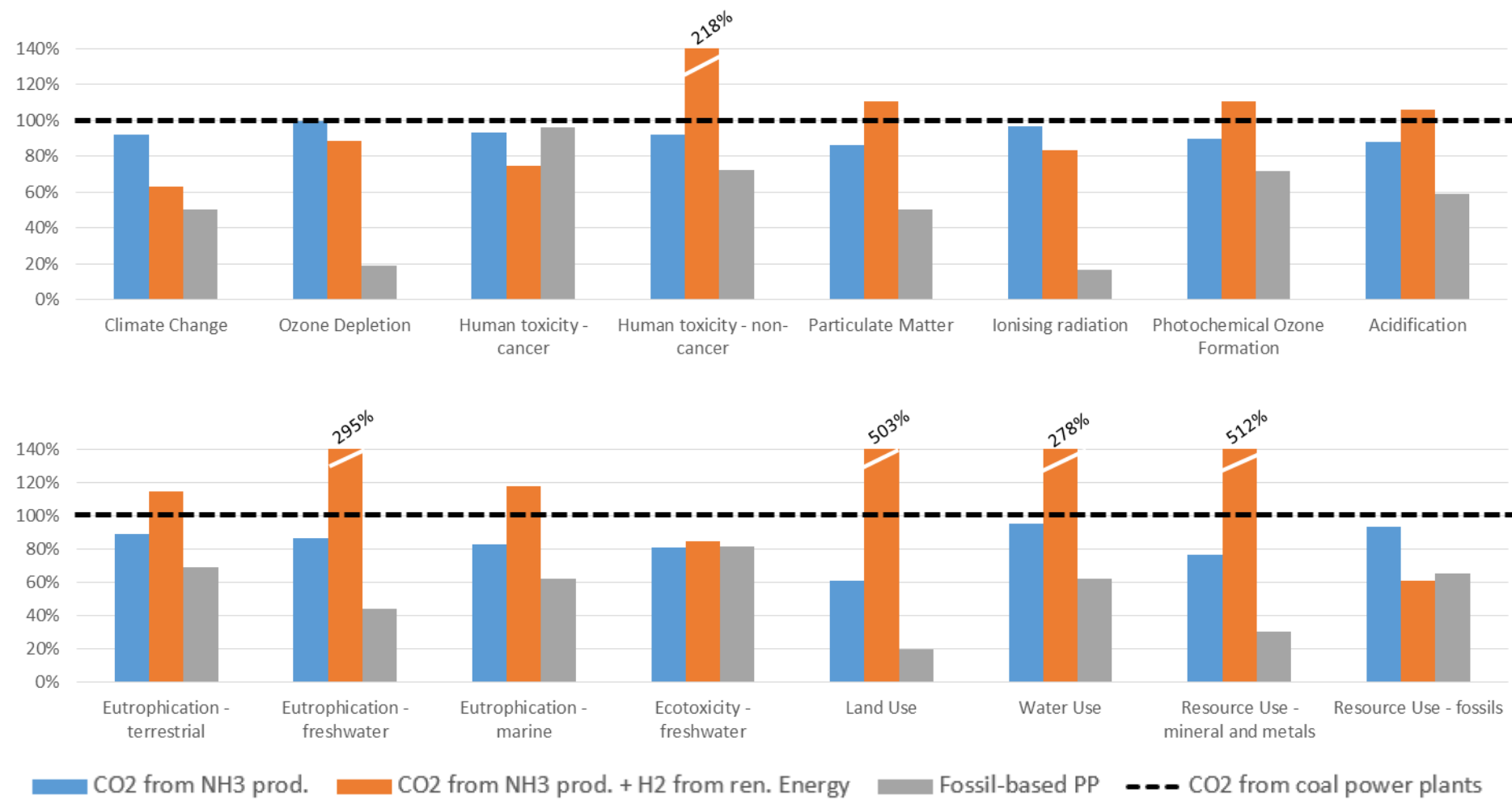
The renewable electricity mix considered as an input to Hydrogen production via water electrolysis (modelled as described in Section 5.4.1.3) was defined based on the most recent electricity generation statistics in Europe from the International Energy Agency (IEA, 2019), and included the following sources: 37.8% hydropower, 33.6% wind power, 16.7% biomass-based electricity, 11.2% photovoltaic, and 0.7% Geothermal. Electricity generation through each of these sources was modelled based on suitable EF-compliant datasets.

Figure 5.21 shows the results of the analysis, expressing recalculated impacts of CO<sub>2</sub>-based PP food packaging film as a function of the impacts of the base case of the same scenario, which is set as a 100% baseline. The impacts of fossil-based PP packaging film are also reported for comparative purposes, and expressed according to the same criteria.

Sourcing CO<sub>2</sub> from Ammonia production plants rather than coal-based power plants decreases the impact of CO<sub>2</sub>-based PP packaging film in several (nine) impact categories, with a reduction between 10% and 23%. In the remaining categories the observed decrease is not significant (lower than 10%). This general improvement is mostly a consequence of the reduced energy (especially heat) consumption for CO<sub>2</sub> supply, which no longer requires the CO<sub>2</sub> extraction process (as discussed above). Despite the observed improvements, CO<sub>2</sub>-based PP film still shows a worse performance compared to its fossil-based counterpart in the vast majority of the impact categories. The only exceptions are represented by Human Toxicity – cancer and Ecotoxicity – freshwater, where both alternatives are comparable.

<sup>81</sup> Consumption of electricity was adjusted to 1.12 MJ/kg CO<sub>2</sub>, including the demand for both purification (0.72 MJ/kg CO<sub>2</sub>; Althaus et al., 2007) and liquefaction (0.4 MJ/kg CO<sub>2</sub>; Von der Assen et al., 2015). Heat consumption, for compression only, was set to 0.01 MJ/kg CO<sub>2</sub>, based on Von der Assen et al. (2015).

1 The additional conversion to electrolytic Hydrogen based on renewable energy from the  
2 current EU mix allows an impact reduction (between 12% and 39%) only in six impact  
3 categories, while in most of the remaining categories an increase is observed. This is  
4 especially the case of Resource Use – minerals and metals (412%), Land Use (403%),  
5 Eutrophication Freshwater (195%), Water Use (178%), and Human Toxicity – non-cancer  
6 (118%), while for the other categories the increase is lower (10-18%). In most  
7 categories, the reason for this worsened performance is the presence of biomass-based  
8 energy in the applied renewable electricity mix (despite its modest share, i.e. 17%). If  
9 the latter is replaced, for instance, by wind power, only three categories show an  
10 increased impact (albeit significant), i.e. Resource Use – minerals and metals (494%),  
11 Water Use (151%), and Eutrophication – freshwater (135%). In all of the remaining  
12 categories an impact reduction between 12% and 42% is instead observed. However,  
13 regardless of the applied energy mix, the overall comparison with fossil-based PP film is  
14 not heavily affected, with the latter still showing a better performance in all impact  
15 categories except for Human Toxicity – cancer, Ecotoxicity – freshwater, Resource Use –  
16 fossils and, when biomass-based electricity is replaced by wind power, Photochemical  
17 Ozone Formation. In these categories, CO<sub>2</sub>-based PP film is comparable with its fossil-  
18 based counterpart, or even preferable in the case of Human Toxicity – cancer and  
19 Ecotoxicity – freshwater (only when biomass-based electricity is removed).  
20



**Figure 5.21.** Results of the sensitivity analysis on the type of CO<sub>2</sub> source and Hydrogen production technology for CO<sub>2</sub>-based PP food packaging film. Scenario impacts are expressed as a percentage of the impacts of the base case of the affected scenario (i.e. CO<sub>2</sub>-based PP food packaging film with CO<sub>2</sub> sourced from coal-based power plants and H<sub>2</sub> produced from the current average mix of technologies), which is set as a baseline. Note that in some impact categories a part of the results is out of scale and is curtailed.

### 5.7.8.6 Modelling of CO<sub>2</sub> as a feedstock for polymer production

This sensitivity analysis explores the application of alternative methodological approaches for the modelling of the use of CO<sub>2</sub> as a feedstock for polymer production, compared to the CFF applied as a base case to CO<sub>2</sub>-based PP food packaging film. The following approaches were considered, focusing on the base case scenario that assumes CO<sub>2</sub> capture from coal-based power plants, and Hydrogen production via the current average mix of technologies:

1. "Cut-off" approach (raw gaseous CO<sub>2</sub> as a waste for recycling);
2. "50:50" waste allocation approach (raw gaseous CO<sub>2</sub> as a waste for recycling);
3. System Expansion via substitution (CO<sub>2</sub>-based Propylene as a co-product of the whole Carbon Capture and Utilisation system).

These approaches are all described in detail in Section 4.4.4 of the Method, and only a brief overview is provided below.

"Cut-off" (or "zero-burden") approach: according to this approach, no burdens from any upstream activity taking place before arising of waste CO<sub>2</sub> are assigned to this feedstock, since supply and conversion of the CO<sub>2</sub>-providing fossil-based feedstock (e.g. coal extraction and combustion in power plants) are totally allocated to the CO<sub>2</sub> source (e.g. electricity production). On the other hand, downstream burdens from disposal (incineration and landfilling) of the CO<sub>2</sub>-based product (e.g. CO<sub>2</sub>-based PP food packaging film), are totally allocated to the life cycle of such product.

"50-50" waste allocation approach: this approach implies that the burdens from upstream activities associated with the supply of the CO<sub>2</sub>-providing fossil-based feedstock (e.g. coal extraction and transport) are equally split between the CO<sub>2</sub> source and the CO<sub>2</sub>-based product. The same criteria applies to the burdens from downstream disposal (incineration and landfilling) of the CO<sub>2</sub>-based product (e.g. PP packaging film).

System Expansion via substitution: in this approach, the Carbon Capture and Utilisation system as a whole is initially considered. This is represented by the coupled production system delivering both the main product of the CO<sub>2</sub> source (e.g. coal-based electricity) and the CO<sub>2</sub>-based product (e.g. Propylene) as simultaneous co-products. The burdens associated with the production of the sole CO<sub>2</sub>-based product are then calculated by subtracting those related to the production of the non-relevant co-product (e.g. electricity) in an uncoupled system without CO<sub>2</sub> capture. This means that electricity from the combined production system is assumed to replace electricity generated in an uncoupled production system where CO<sub>2</sub> capture is not performed. In practical terms, this approach implies modelling a system similar to that resulting from applying the "cut-off" approach, which in this case means a system beginning with raw CO<sub>2</sub> capture from flue gas of the power plant and including all downstream supply chain processes in the life cycle of the CO<sub>2</sub>-based product (i.e. PP food packaging film). However, the system is also credited with an avoided release of (fossil) CO<sub>2</sub> corresponding to the amount taken up in the CO<sub>2</sub>-based product (although physically representing the release that would take place in the uncoupled electricity production process without CO<sub>2</sub> capture).

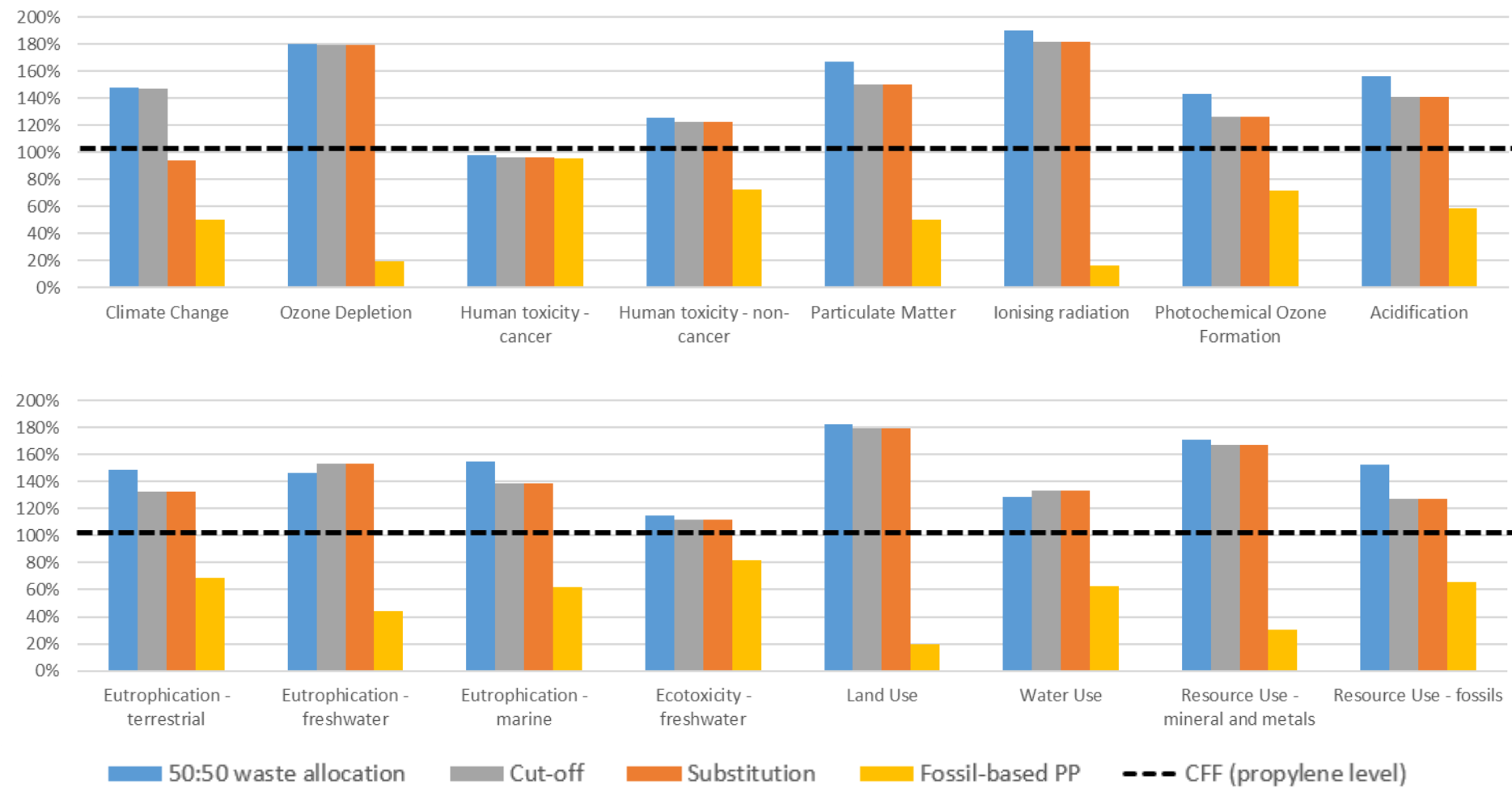
The results of the sensitivity analysis are shown in Figure 5.22, where the impacts of CO<sub>2</sub>-based PP packaging film calculated by applying the three alternative modelling approaches are expressed as a function of the impacts of the base case of the same scenario (where the CFF is applied), which is set as 100% baseline. The impacts of fossil-based PP packaging film are also reported for comparative purposes, and expressed with the same criteria. Moreover, Figure 5.23 provides a more detailed focus on the results obtained for the two most relevant impact categories identified in Section 5.7.1 (i.e. Climate Change and Resource Use – fossils).

All the explored alternative approaches involve an increase in the potential impact of CO<sub>2</sub>-based PP food packaging film, compared to the application of the CFF, in the majority of the impact categories. Exceptions are the categories of Human Toxicity –

1 cancer and Ecotoxicity – freshwater, where all the examined approaches (including the  
2 CFF) result in comparable impacts. Similarly, in the case of Climate Change, System  
3 Expansion via substitution is comparable to the application of the CFF (in contrast to the  
4 other alternative approaches, which both provide an increased impact). Indeed, in the  
5 case of substitution, the potential impact increase is mitigated by CO<sub>2</sub> emissions savings  
6 from avoided electricity generation in a plant without CO<sub>2</sub> capture.

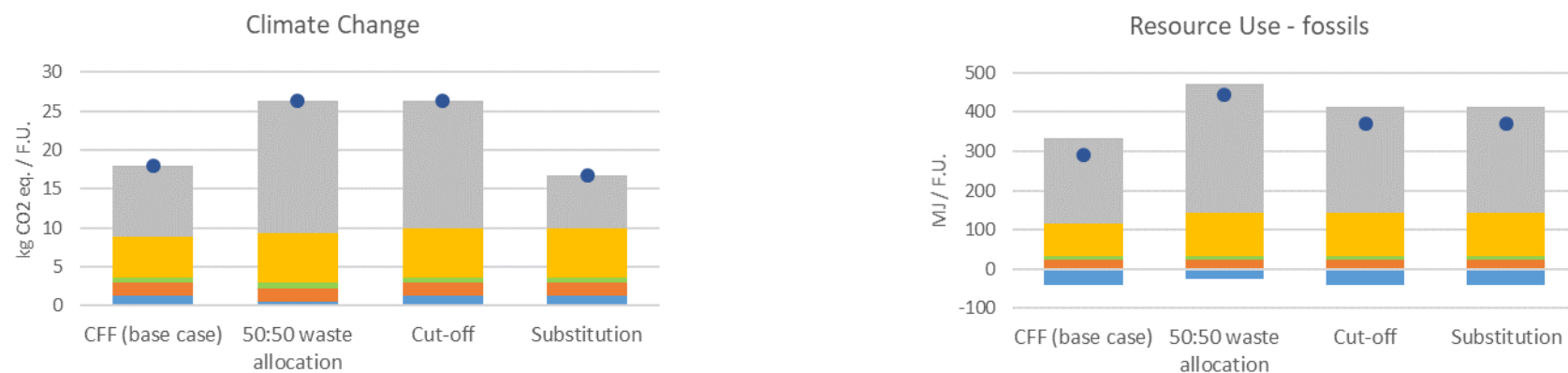
7 Leaving these few exceptions apart, the main reason for the increased impacts from  
8 applying the three alternatives approaches to the CFF is in that the latter implies  
9 modelling CO<sub>2</sub>-based Propylene as being partly produced via the conventional, fossil-  
10 based synthesis route (i.e. naphtha cracking), which is currently less impacting than the  
11 CO<sub>2</sub>-based route (i.e. the process chain from CO<sub>2</sub> capture to Propylene production).  
12 Conversely, only the CO<sub>2</sub>-based route to Propylene synthesis is modelled in the explored  
13 alternative approaches, leading to the observed impacts increase. The increase is  
14 identical when applying both the Cut-off and the Substitution approach (if Climate  
15 Change is excluded, as discussed above). In several categories, also the 50-50 waste  
16 allocation approach involves a comparable increase to these two approaches, while in the  
17 other categories the increase is generally larger (Particulate Matter, Photochemical Ozone  
18 Formation, Acidification, Eutrophication – terrestrial, Eutrophication – marine, and  
19 Resource Use – fossils). This is because, in the 50-50 approach, the CO<sub>2</sub>-based product  
20 carries also a share of the burdens from the supply of the original CO<sub>2</sub>-providing fossil  
21 feedstock (i.e. coal), which compensate for the discounted share of burdens from final  
22 disposal of the CO<sub>2</sub>-based product.

23



**Figure 5.22.** Results of the sensitivity analysis on the approach applied to the modelling of the use of captured CO<sub>2</sub> as a feedstock for CO<sub>2</sub>-based PP food packaging film production. Scenario impacts are expressed as a percentage of the impacts of the base case of the CO<sub>2</sub>-based scenario, which is set as a baseline. In this scenario, the CFF is applied as a modelling approach, CO<sub>2</sub> is sourced from coal-based power plants and Hydrogen comes from the current average mix of production technologies.





Life Cycle stage	CFF (base case)	50:50 waste allocation	Cut-off	Substitution
Feedstock supply	9.1	17.1	16.3	6.8
Polymer production	5.2	6.4	6.4	6.4
Article production	0.7	0.7	0.7	0.7
Distribution	1.7	1.7	1.7	1.7
End of Life	1.2	0.5	1.2	1.2
Total	17.9	26.4	26.3	16.8

Life cycle stage	CFF (base case)	50:50 waste allocation	Cut-off	Substitution
Feedstock supply	217	329	269	269
Polymer production	81.6	109	109	109
Article production	11.2	11.2	11.2	11.2
Distribution	22.6	22.6	22.6	22.6
End of Life	-41.2	-26.5	-41.2	-41.2
Total	291	445	371	371

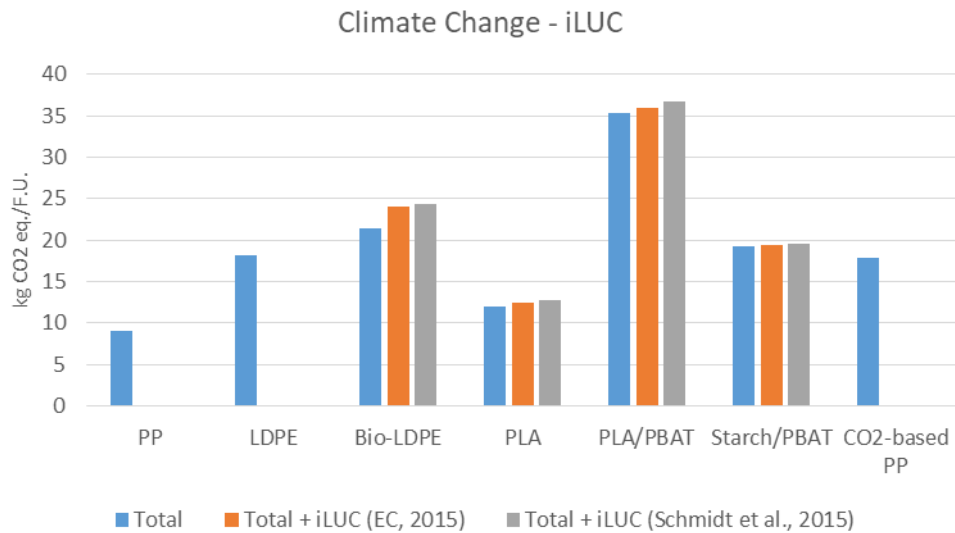
■ Feedstock Supply ■ Polymer Production ■ Article Production ■ Distribution ■ End of Life ● Total

**Figure 5.23.** Detailed results of the sensitivity analysis on the approach applied to the modelling of the use of CO<sub>2</sub> as a feedstock for PP food packaging film production in the most relevant impact categories (Climate Change and Resource Use – fossils).

5.7.8.7 iLUC model and factors

This sensitivity analysis explores the use of an alternative model (i.e. Schmidt et al., 2015) to quantify the iLUC contribution to the Climate Change impact of bio-based food packaging films (i.e. Bio-LDPE, PLA, Starch/PBAT and PLA/PBAT films). The iLUC factors calculated through the mentioned model were applied in place of the GHG emission factors from EC (2015) applied as a base case (Table 5.13). The results are displayed in Figure 5.24, limited to the Climate Change category, which is the only one affected by this sensitivity analysis.

Overall, the application of alternative iLUC factors only marginally affects the results. The highest impact increase observed when applying the EC (2015) factors was equal to 12% (sugarcane-based Bio-LDPE packaging film), while applying the model from Schmidt et al. (2015) leads to a comparable increase of 13% (corresponding to an 8% increase in relative terms). For PLA and PLA/PBAT packaging films (which rely on maize as a feedstock) the observed increase is higher, with an iLUC contribution which is more than doubled (+111%) compared to the base case assessment. Despite this large relative variation, the iLUC contribution to the overall lifecycle impact of these alternatives is still limited, increasing from 2.8% to 5.8% for PLA film and from 1.7% to 3.6% for PLA/PBAT film. A similar relative increase is also observed for Starch/PBAT packaging film (+114%), which is still based on maize (and other starch crops) as a feedstock (although limited to 40% of the polymer blend). This alternative was the one showing the lowest iLUC contribution in the base case results (0.4%), which is increased to 0.9% when the model from Schmidt et al. (2015) is applied, again with no relevant consequences on the overall Climate Change impact. In conclusion, the results of this case study can be considered reasonably robust with respect to the estimated iLUC contribution (which is generally modest or negligible), and the comparison among the affected packaging film scenarios ant the respective fossil-based scenario of references is not affected by the application of alternative factors.



**Figure 5.24.** Results of the sensitivity analysis on the applied model to quantify the iLUC contribution to the total Climate Change impact of bio-based food packaging films.

## 6 Case study 3: Trays for food

This case study focuses on single-use trays used for packaging of food products such as fresh fruit and vegetables, cold meat, and bakery products. In particular, the focus is on generic trays with arbitrarily fixed dimensions (334 x 274 x 21 mm) and a variable wall thickness, to ensure a comparable (barrier) performance with the use of different materials. This allows to explore a wider range of materials and feedstock sources within the case study, as well as to avoid unnecessarily restricting its scope to a specific food item with the related packaging requirements. Rather, the study can be considered to inherently cover all those food items compatible with the permeability features/performance considered as a reference (as better detailed in the rest of this section).

### 6.1 Assessed scenarios

A number of scenarios were analysed to explore the potential impacts associated with the use of alternative feedstock sources or materials for the manufacturing of trays for food (Table 6.1). Polymers traditionally used for trays production are PET and PP from (virgin) fossil-based feedstock sources, which were considered as reference materials for the comparison (Scenarios 1 and 2). High impact polystyrene (HI-PS), oriented PS, and expanded PS (EPS) are also used, but their application in the market appears more limited compared to the previous materials (although no -official- data in this respect are available). Moreover, the use of alternative feedstock sources is currently more restricted for these polymers, since only very first examples of the use of recycled EPS (mostly for non-food grade applications) are available, and relevant life cycle inventory data in this respect are very scarce. Therefore, none of these materials was considered in this study, also to keep a reasonable and balanced number of reference and alternative scenarios for the assessment.

The use of recycled PET from separately collected, post-consumer plastic waste was then explored as a feedstock for trays made of such polymer (Scenario 3). An estimate of the current average recycled content in PET bottles at the EU level was considered as a base case (i.e. 24%<sup>82</sup>), in the absence of specific data for trays. However, since it is technically feasible to incorporate higher shares of recycled polymer in PET trays production (virtually up to 100%), the use of a 100% recycled content was also explored as a sensitivity analysis. This allows to assess the effects of a complete substitution of the virgin material by its recycled counterpart, and to quantify the maximum potential environmental benefits or drawbacks expected from this replacement.

A partially bio-based, drop-in alternative to fossil-based PET is also 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) was explored in Scenario 4, considering Brazilian sugarcane as a feedstock for bio-based MEG. This choice is in line with the origin of bioethanol currently used by the largest bio-MEG supplier worldwide, which is located in India and relies on bioethanol sourced in Brazil.

Three biodegradable alternative polymers (totally or partially bio-based) were also assessed for food trays production. These include: Polylactic Acid (PLA, fully bio-based, Scenario 5)<sup>83</sup>, a blend of Thermoplastic Starch (TPS) and PBAT<sup>84</sup> (partially-bio-based, Scenario 6), referred to as Starch-based polymer or Starch blend in the following, and bio-based Polybutylene Succinate (Bio-PBS, fully bio-based, Scenario 7). These materials are characterised by an oxygen transmission rate similar to that of PET (having the same order of magnitude), and hence represent a possible alternative to the use of this

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<sup>82</sup> For more information on how this estimate was derived, please refer to Section 4.1 (Beverage bottles case study).

<sup>83</sup> Bi-axially oriented PLA is specifically considered, according to the features of PLA packaging found on the market that make use of this material in a “pure” form.

<sup>84</sup> PBAT: Polybutylene Adipate co-Terephthalate.

polymer in food packaging applications. Bio-based PP was also investigated (Scenario 8), although it was predicted to be available on the market only in 2023 (according to the latest official data available when developing this case study; EUBP, 2019). Sugarcane grown in Brazil was again considered as a feedstock for bio-based Propylene production, being the one most widely used at present for bioethanol-derived polymers.

For PLA production, US maize was considered as a feedstock, since nearly 90% of PLA currently consumed in Europe comes from the US (Eurostat, 2019a), where maize is used as a source of starch-derived sugar (dextrose) by the largest producer in the country (Vink and Davies, 2015). US maize was also considered as a feedstock for both Bio-PBS co-monomers (Butanediol and Succinic Acid). Conversely, starch used for TPS production was derived from the current EU-average mix of starch crops, being one of the major producers of starch-based polymers located in Europe. The mix includes maize (47% on starch basis), wheat (40%) and potatoes (13%) (Starch Europe, 2019). PBAT used as a blending copolymer with TPS in starch-based trays is based on fossil sources, and no agricultural feedstock is hence needed in this case.

It must be noted that different scenarios may rely on processes having different technology readiness levels (TRLs). In particular, manufacturing of Bio-PBS is based on a relatively new technology compared to the manufacturing of PET, which relies on a well-established technology. Therefore, the results related to this alternative need to be interpreted with more caution.

Regarding End of Life, all currently viable treatment and disposal options for trays made of each specific material were considered. For trays made of materials not yet widely established on the market (e.g. biodegradable polymers), potentially viable options as of today were considered, taking into account relevant material properties (e.g. biodegradability), and the options currently applied to trays made of the material(s) they intend to replace. Therefore, for trays consisting of conventional, non-biodegradable polymers (i.e. PET and PP, from any feedstock) viable options that were considered include mechanical recycling, incineration and landfilling. For trays consisting of biodegradable polymers (i.e. PLA, Starch/PBAT and Bio-PBS) biological treatment options were also considered to be viable (i.e. composting and anaerobic digestion). However, Bio-PBS trays were not considered to undergo anaerobic digestion due to the demonstrated low degradability of this polymer under anaerobic conditions (UBA, 2018). Moreover, such options are not applicable to Bio-PET and Bio-PP trays, which despite being partially bio-based are not biodegradable, having the same characteristics of their fossil-based counterpart ("drop-in" solutions).

As a base case, the impacts of each trays for food scenario were assessed with reference to an EU-average End of Life scenario including all the options currently applied (or which would be potentially applied) at the EU level to trays made of the specific material. These include all the viable options reported above, which were combined as described in Section 6.4.5.1 on End of Life modelling. In addition, scenario impacts were calculated by individually considering the application of each currently viable (or potentially viable) End of Life option.

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**Table 6.1.** LCA scenarios assessed for the trays for food case study.

Scenario	Polymer	Monomer or Co-polymer	Feedstock	End of Life options <sup>(1)</sup>
<i>S1 - Conventional polymer 1</i>	PET	MEG <sup>(2)</sup> PTA <sup>(3)</sup>	Fossil-based (crude oil/natural gas)	Recycling Incineration Landfilling
<i>S2 - Conventional polymer 2</i>	PP	Propylene	Fossil-based (crude oil/natural gas)	Recycling Incineration Landfilling
<i>S3 - Alternative polymer 1</i>	R-PET (24% recycled content)	MEG <sup>(2)</sup> PTA <sup>(3)</sup>	Waste PET (post-consumer)	Recycling Incineration Landfilling
<i>S4 - Alternative polymer 2</i>	Bio-PET	Bio-MEG <sup>(2)</sup> PTA <sup>(3)</sup>	Sugarcane (BR) Crude oil/natural gas	Recycling Incineration Landfilling
<i>S5 - Alternative polymer 3</i>	PLA (Bi-axially oriented)	Lactic Acid	Maize (US)	Recycling Composting Anaerobic digestion Incineration Landfilling
<i>S6 - Alternative polymer 4</i>	Starch/PBAT blend <sup>(4)</sup>	Starch 1,4-Butanediol Adipic acid PTA <sup>(3)</sup>	EU mix of starch crops <sup>(5)</sup> Crude oil/natural gas	Recycling Composting Anaerobic digestion Incineration Landfilling
<i>S7 - Alternative polymer 5</i>	Bio-PBS	Bio-Succinic Acid Bio-BDO <sup>(6)</sup>	Maize (USA)	Recycling Composting Incineration Landfilling

<sup>(1)</sup> The impacts of scenarios were individually assessed for each listed End of Life option, as well as for a combination of such options reflecting as far as possible the current (real or potential) average situation at the EU level.

<sup>(2)</sup> MEG: Mono Ethylene Glycol.

<sup>(3)</sup> PTA: Purified Terephthalic Acid.

<sup>(4)</sup> A 40% Starch and 60% PBAT blend is considered, according to the information received from stakeholders' consultation, reporting that "renewability" of starch-based polymers used in packaging applications can reach 40%.

<sup>(5)</sup> Maize (47%), Wheat (40%), Potatoes (13%), in terms of starch product equivalents (Starch Europe, 2019).

<sup>(6)</sup> BDO: 1,4-Butanediol.

## 6.2 Functional Unit and reference flow

The main function of the studied article is to deliver packaged food from producers to final users. The functional unit was thus defined as "*delivering packaged food in the EU by means of 1000 single-use trays with dimensions 334x274x21 mm, without breaking during transport and handling, and ensuring a comparable shelf life of the packaged product*"<sup>85</sup> (Table 6.2).

<sup>85</sup> It is assumed that the oxygen permeability is the only parameter affecting the product's shelf life. Therefore, tray materials with similar Oxygen Transmission Rates (OTR) will ensure comparable shelf lives.

In this study, it was assumed that the shelf life is uniquely determined by the oxygen permeability/barrier of trays, reflecting the Oxygen Transmission Rate (OTR) of the respective material. The ability of trays of not breaking due to bending (i.e. the most important stress that trays may undergo) can instead be assumed to mainly depend on the flexural/bending strength of such materials (beyond their thickness in trays).

The different compared materials have different technical properties (e.g. mechanical and permeability properties) for a given tray wall thickness, affecting their ability to achieve a defined performance/functionality requirement (or design constraint) such as a given flexural strength or oxygen barrier. These differences in technical properties may thus ultimately affect the mass of material needed to fulfil the functional unit (i.e. the reference flow), as far as the specified performance requirements can be met by adjusting the thickness of trays walls, while maintaining their dimensions. Depending on the specific performance/functionality requirement that is considered (or design constraint that needs to be prioritised), a different tray wall thickness, and hence material mass, is generally needed. Therefore, a limiting requirement or constraint can be identified in the one implying the highest thickness (and mass) of material, which ensures that also the remaining requirements are met, provided that these do not counteract each other (i.e. if a thicker tray wall does not imply a worsened performance towards any other requirement).

For illustrative purposes, in this case study it was assumed that a single performance requirement could be prioritised at the time (regardless of whether the other one is met or not), to independently evaluate how the comparative performances of the assessed scenarios are affected by each specific material property. As a base case, the need of ensuring a comparable shelf life (and hence oxygen barrier/permeability) was assumed as the performance requirement to be prioritised (or the limiting design constraint). In particular, the average oxygen permeability of (oriented) PET trays with a thickness of 1 mm was assumed as the reference performance to be achieved, corresponding to an OTR equal to 62 cm<sup>3</sup>/m<sup>2</sup>-day (23°C, 0% RH), according to PolyPrint (2019).

**Table 6.2.** Definition of the functional unit for trays for food LCA scenarios.

Aspect	Description
<b>"What"</b> (function(s) or service(s) provided)	Delivery of packaged food by means of single-use trays with dimensions 334x274x21 mm
<b>"How much"</b> (extent of the function(s) or service(s))	1000 single-use trays
<b>"How well"</b> (expected level of quality)	Without breaking during transport and handling (due to bending), and ensuring a comparable product shelf life
<b>"How long"</b> (duration/lifetime of the function or service)	One time
<b>"Where"</b> (location/geography of the service)	In the EU

No immediate deterministic relationships are available to relate the oxygen transmission rate (or other technical properties) of a material, with the corresponding thickness (and hence with the mass) needed to achieve a required (barrier) performance. Therefore, to calculate the reference flow (i.e. the amount of polymer required to fulfil the functional unit), accounting for differences in relevant technical properties of the compared materials, the concept of Material Substitution Factors (MSFs) was applied. As already discussed in Section 5.2, MSFs express the ratio between the mass of a given material and of a reference material, needed to perform identically a given technical function, and can be calculated on the basis of so-called Material Indices (MIs). MIs were introduced by

Ashby (1999) to facilitate initial material selection in mechanical design, and aims to score the performance of a material with respect to a given (mechanical) function/performance (e.g. strength or stiffness), based on relevant material properties (e.g. tensile strength or Young's modulus and density). Materials with the highest Material Index can perform the intended function with the minimum mass<sup>86</sup>. Therefore the ratio between Material Indices of the reference material and of an alternative one, can be considered to represent the Material Substitution Factor, according to the following Equation:

$$MSF = m/m_{ref} = MI_{ref}/MI$$

where  $m$  and  $m_{ref}$  respectively represent the mass of the alternative and of the reference material needed to perform the intended function, while  $MI$  and  $MI_{ref}$  represent the Material Indices of the same materials, with respect to such function.

Material Indices were calculated following the same philosophy adopted by Ashby (1999) when considering mechanical functions and the related material properties as a reference (no specific expressions are defined by the author to score material performances towards gas permeability). Considering that both a reduced oxygen transmission rate and density concur to either an improved material performance or to a reduced material mass over a given extension and thickness, MIs were thus quantified as the reciprocal of the product between the two mentioned material properties (i.e.  $MI = (OTR \cdot \rho)^{-1}$ ). The main calculations carried out for this purpose, and the values assumed for relevant material properties are summarised in Table 6.3, considering (oriented) PET as a reference material (as discussed above).

Note that, due to the high OTR of (oriented) polypropylene (PP) compared to the rest of the polymers (1550-2500 cm<sup>3</sup>/m<sup>2</sup>-day), trays made of PP were not directly compared with those made of the other investigated materials. It was therefore assumed that PP trays have a different function in the market, and were only compared among different feedstock sources (i.e. virgin, recycled and bio-based PP). The mass and the reference flow of PP trays was hence calculated by assuming a default thickness of 1 mm, as in the case of PET trays considered as a reference.

It is also noted that the purpose here is to illustrate the concept of integrating relevant material properties in the calculation of the reference flow, for a more appropriate, performance-based comparative assessment. In this perspective, the study does not aim at providing accurate estimates of the potential impacts of each food tray material when a given technical performance needs to be ensured. Therefore, the calculation approach adopted in this study is considered suitable, despite it provides estimates of material requirements that may not be necessarily reflected in real products on the market.

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<sup>86</sup> Note, however, that material indices do not express the mass of material needed to perform a given (mechanical) function, but only the performance of the material with respect to that function.

**Table 6.3.** Calculation of the reference flow for trays for food LCA scenarios, based on material substitution factors.

Polymer	Density - $\rho$ (kg/m <sup>3</sup> )	OTR <sup>(1)</sup> (cm <sup>3</sup> /m <sup>2</sup> /d)	MI (OTR* $\rho$ ) <sup>-1</sup>	MSF (m/m <sub>ref</sub> )	Reference flow (kg/FU)	Thick-ness (mm) <sup>(2)</sup>
PET (all types of feedstock)	1380 <sup>(3)</sup>	62.0 <sup>(4)</sup>	0.0449	1	165.01 <sup>(5)</sup>	1.00
PP	908 <sup>(6)</sup>	2025 <sup>(4)</sup>	-	-	107.61 <sup>(7)</sup>	1.00
PLA (bi-axially oriented)	1,240 <sup>(8)</sup>	22.9 <sup>(9)</sup>	0.0185	0.37	60.95	0.41
TPS/PBAT blend	1,290 <sup>(10)</sup>	63.0 <sup>(11)</sup>	0.0488	1.02	167.67	1.10
Bio-PBS	1,240 <sup>(12)</sup>	35.0 <sup>(12)</sup>	0.0282	0.56	93.15	0.63

<sup>(1)</sup> OTR: Oxygen Transmission Rate. OTR values are given at 23°C, 0% Relative Humidity and for 1 mm material thickness.

<sup>(2)</sup> The thickness of PET trays (reference material) was assumed (and used to calculate the reference flow, see note 5). For trays made of the other materials, the thickness was backward calculated for checking purposes, based on the estimated reference flow and the assumed tray dimensions.

<sup>(3)</sup> Omnexus (2019).

<sup>(4)</sup> PolyPrint (2019). OTR of PP trays is not taken into account in the calculation of the reference flow.

<sup>(5)</sup> For PET trays (reference material), the reference flow was calculated assuming a tray thickness of 1 mm, corresponding to the thickness for which the assumed OTR value is determined.

<sup>(6)</sup> Average of the density range reported by PlasticsEurope (2020), i.e. 895-920 kg/m<sup>3</sup>.

<sup>(7)</sup> For PP trays the reference flow was not calculated based on Material Substitution Factors (as discussed above in this section), but based on tray dimensions and assuming a wall thickness of 1 mm as of for PET trays considered as a reference.

<sup>(8)</sup> Average of values from the technical specifications of two BO-PLA packaging film available on the market.

<sup>(9)</sup> Armentano et al. (2015).

<sup>(10)</sup> Weighted average of the densities of the two constituting copolymers (Jost, 2018), considering the assumed polymer composition (i.e. 40% TPS and 60% PBAT).

<sup>(11)</sup> Based on the technical specifications of a starch-based polymer grade suitable for injection moulded packaging items and tableware.

<sup>(12)</sup> MCC Biochem (2019).

## 6.3 System boundary

In all scenarios, the system boundary was set in order to cover the most relevant stages and processes of the full product life cycle (cradle-to-grave perspective), as described below and depicted in Figure 6.1 to 6.7:

- *Feedstock Supply*<sup>87</sup> – covering extraction, transport and possible refining of crude oil and natural gas (fossil-based polymers), collection, transport and sorting of plastic waste (recycled polymers), crop cultivation (bio-based polymers), as well as transport of these feedstock sources to downstream conversion processes (e.g. naphtha cracking, polymer recycling, sugarcane fermentation, wet milling of starch crops);
- *Polymer Production*<sup>88</sup> – covering all the activities associated with the conversion of feedstock materials into the relevant monomer(s) and final polymer, including any transport among these activities and final transport of polymer granulate to downstream manufacturing processes;

<sup>87</sup> Corresponding to the default stage of "Raw Material Acquisition and Pre-Processing" specified in the Method and in the PEF framework.

<sup>88</sup> Corresponding to the default stage of "Raw Material Acquisition and Pre-Processing" specified in the Method and in the PEF framework.



- *Article Production*<sup>89</sup> – including trays manufacturing through injection moulding of polymer granulates, and their transport to the food packing site;
- *Distribution* – including transport of trays (as part of a final packaging item) from the food packing site to retailers and from these to final consumers<sup>90</sup>;
- *End of Life* – covering collection, transport, recycling, biological treatment, incineration, or disposal of trays after use, including any avoided processes from virgin material or energy substitution.

A different nomenclature was applied for some life cycle stages compared to the default nomenclature specified in the Method (and in the PEF framework) to make it more relevant for the investigated supply chains and the project scope. Moreover, the default “Raw Material Acquisition and Pre-processing” stage was further split into two separate sub-stages (i.e. Feedstock Supply and Polymer Production), to allow disaggregating the impacts of feedstock supply from downstream conversion processes, and hence to better appreciate any differences among the use of different feedstock sources (in line with the project goal).

The food packing process (e.g. filling and sealing/wrapping) and the Use Stage were excluded from the assessment, since the study focuses on trays for food delivery and not on the life cycle of a specific packaged food, or of a specific packaging item based on trays (e.g. lidded or wrapped trays). Moreover, the burdens of the mentioned activities and of the Use Stages can be considered identical in all the investigated scenarios, since using different tray materials or feedstock sources for these do not usually affect the burdens of the packing process (e.g. energy consumption) or of the Use stage (e.g. refrigeration requirements or other relevant aspects)<sup>91</sup>. Therefore, such activities can be excluded from a comparative assessment, like the one performed in this study, and their exclusion do not affect the outcome of the comparison among the different scenarios.

Similarly, the life cycle of any additional packaging component that may be used along with trays (e.g. lids, film cover, labels, etc.) was excluded, considering the specific focus of the study on trays. Moreover, it can be reasonably assumed that the same additional components are used, regardless of the material or feedstock applied for trays manufacturing<sup>92</sup>, so that the comparison among the different scenarios is not affected by this omission.

It has to be noted, however, that the mentioned exclusions may weaken the assessment if they are not adequately justified (e.g. if relevant differences exist among compared alternatives, which are not taken into account). Therefore, the Use Stage, food packing activities (and the life cycle of any additional packaging components) shall always be taken in full consideration for possible inclusion in a comparative LCA of trays for food (or tray-based packaging items), paying particular attention to any differences among the compared alternatives. Moreover, such stages shall always be included in LCA studies of packaged food products, in line with system boundary requirements in the Method and in the PEF framework.

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 food trays, of the examined polymers, and of plastics in general, as well as on their release

<sup>89</sup> Corresponding to the default stage of “Manufacturing” specified in the Method and in the PEF framework.

<sup>90</sup> While filled trays are transported in reality (with the respective product content), transport of packaged food was excluded in this case study, due to the exclusive focus on the trays life cycle, and not on the life cycle of a specific packaged food product.

<sup>91</sup> This is true as long as different packaging materials do not imply significantly different product (shelf) lives, which may affect the storage duration (with the respective energy requirements, in case refrigeration is needed) as well as the potential generation of food waste at consumer level. In this situation, the Use Stage should be taken into account, although changes in food waste generation due to a different shelf life is hardly quantifiable. However, in this case study, the requirement to ensure a comparable shelf life in all the compared scenarios inherently excludes this possibility.

<sup>92</sup> Unless a packaging item with specific properties (e.g. biodegradability) is to be produced, so that materials with suitable characteristics would need to be used.

1 and fate over the product life cycle. One exception is represented by additives (mainly  
2 plasticisers) used in Thermoplastic Starch production, which represent a non-negligible  
3 and essential portion of the final polymer (i.e. 25%) and are required to ensure suitable  
4 material properties and processability. Moreover, production of additives used in starch-  
5 based polymers can account for a non-negligible portion of the respective cradle-to-gate  
6 Climate Change impact and energy demand, reaching 46% for polymer grades including  
7 larger shares of additives in the range of 30% (Broeren et al., 2017). Leaving this  
8 exception apart, the exclusion of additives is acknowledged as a limitation of this study,  
9 since additives can also be relevant at the End of Life stage, where they can be released,  
10 as such or after degradation/conversion into different compound(s), in the environment.

11

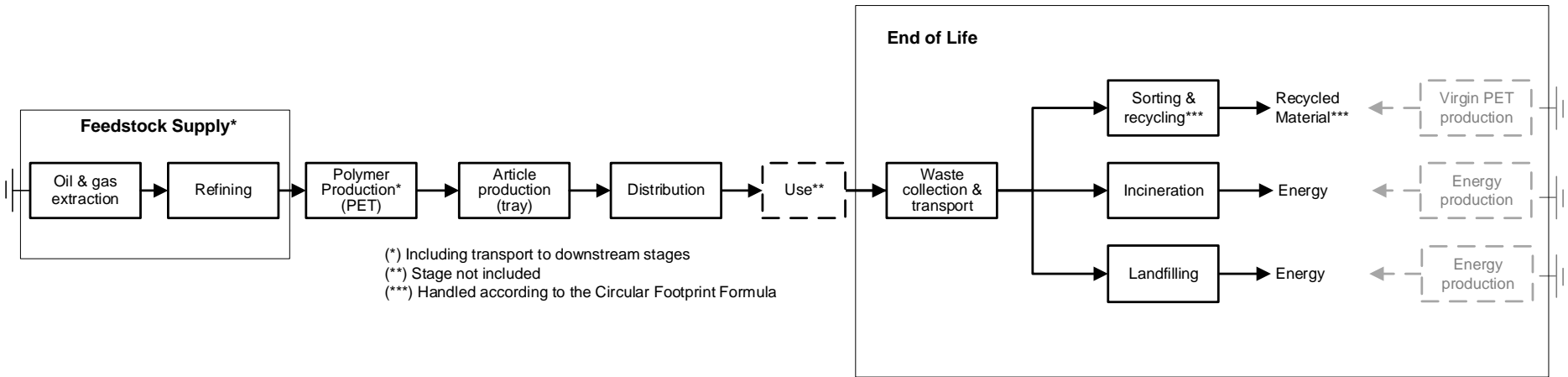


Figure 6.1. System boundary for fossil-based PET trays for food (Scenario 1).

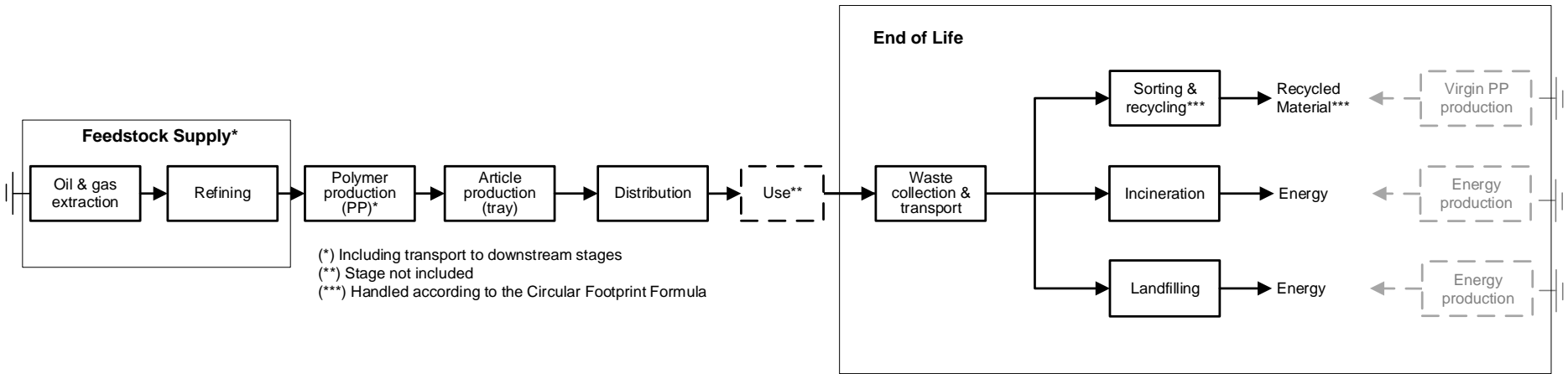
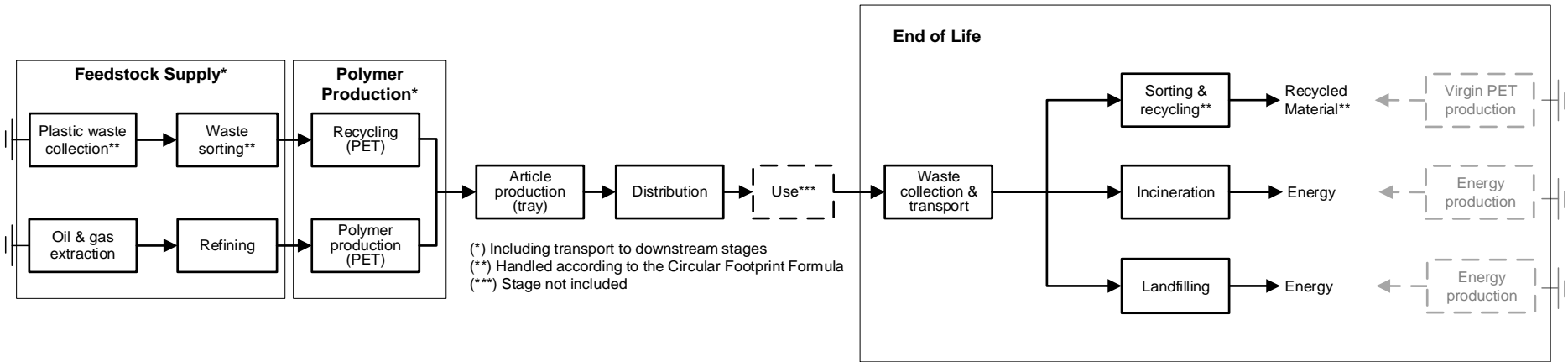
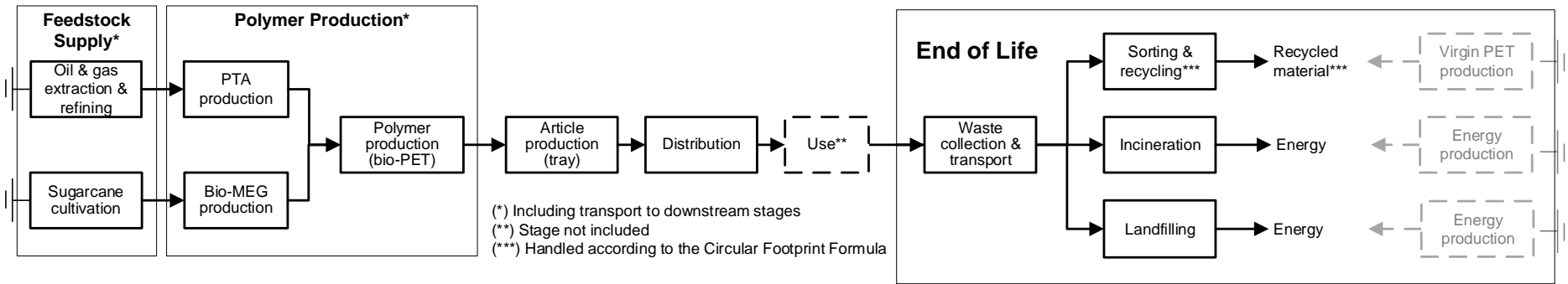


Figure 6.2. System boundary for fossil-based PP trays for food (Scenario 2).



4 **Figure 6.3.** System boundary for 24% recycled PET trays for food (Scenario 3).



8 **Figure 6.4.** System boundary for bio-based PET trays for food (Scenario 4).

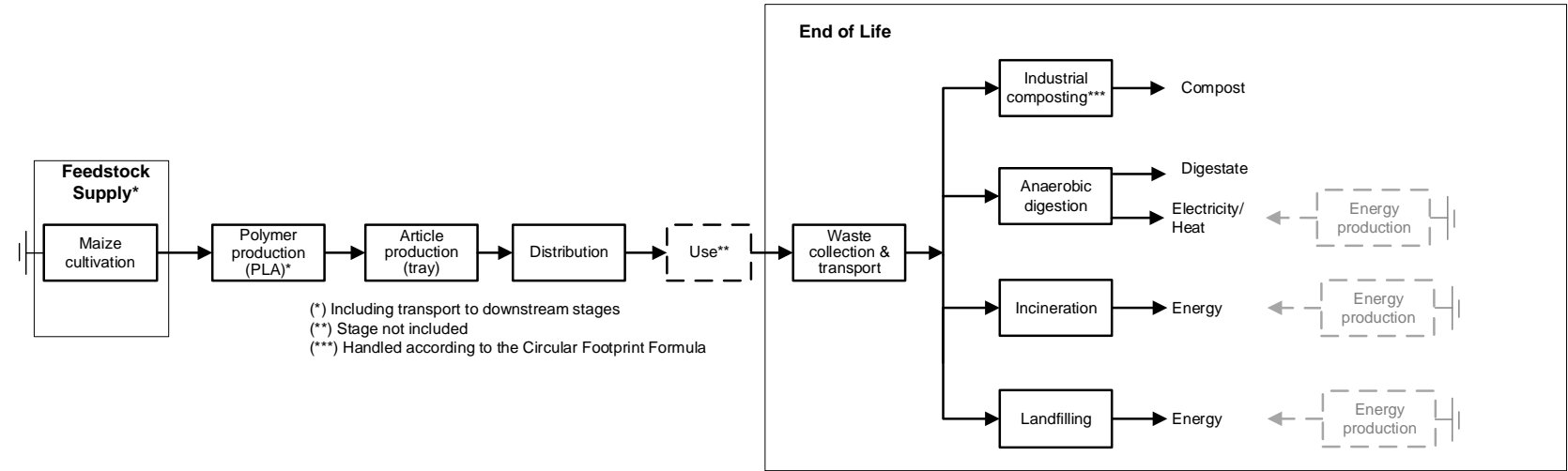


Figure 6.5. System boundary for PLA trays for food (Scenario 5).

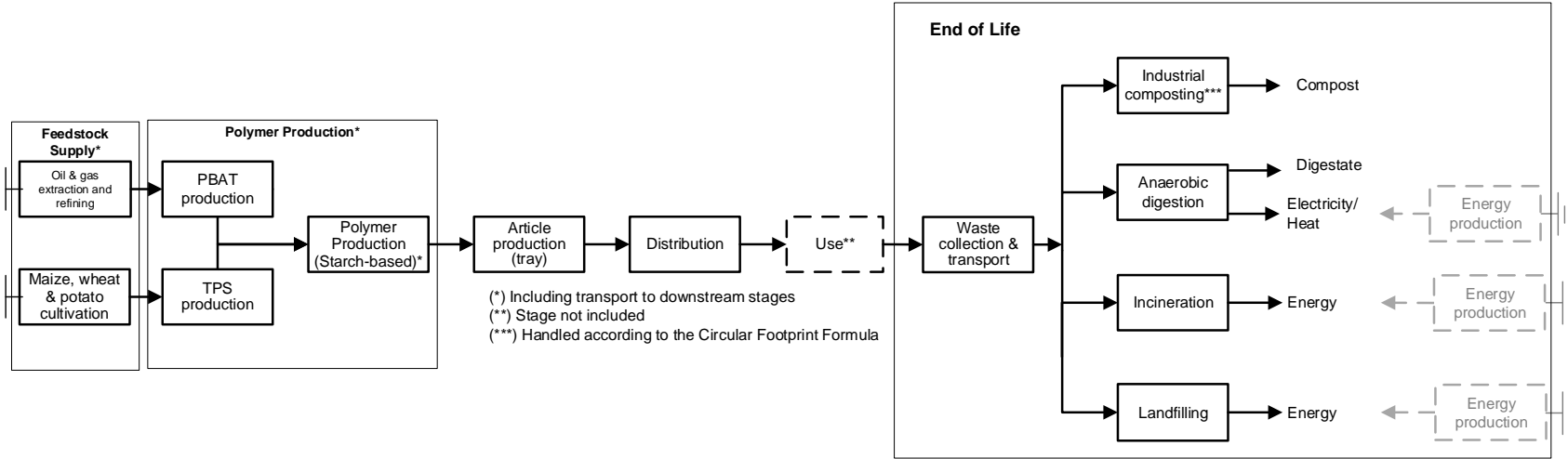


Figure 6.6. System boundary for starch-based trays for food (Scenario 6).

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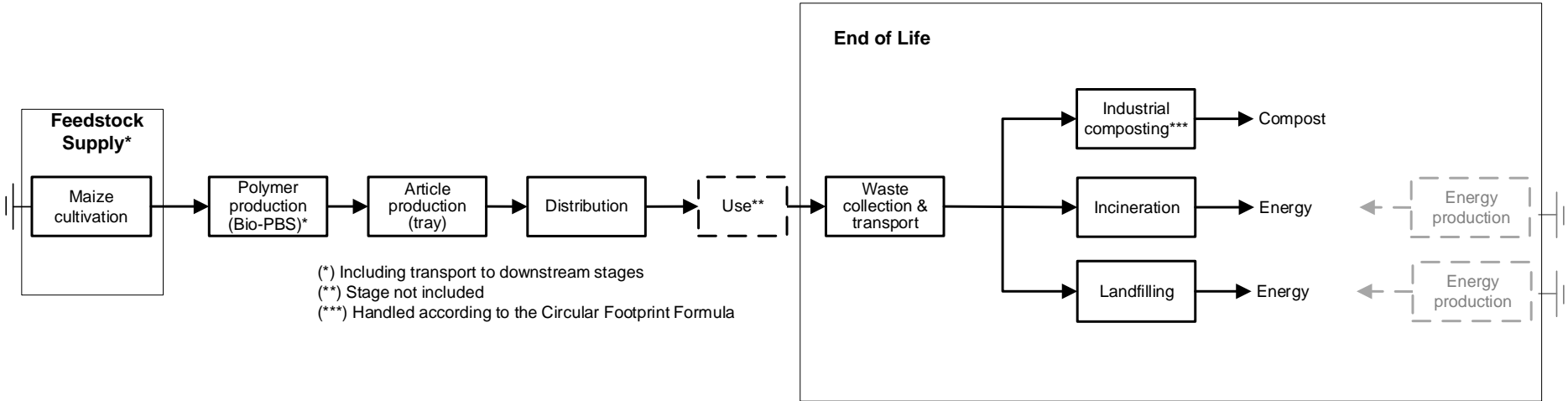


Figure 6.7. System boundary for bio-based PBS trays for food (Scenario 7)

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4  
5

## 6.4 Life Cycle Inventory

This section describes the overall approach for developing the Life Cycle Inventory of the analysed scenarios, along with the related assumptions and data sources. The description is separated by major lifecycle stages, in the following sub-sections (6.4.1 – 6.4.6). The list of processes, related data sources, and main modelling details are provided in Tables B.3.1 to B.3.7 in Annex B.3.

### 6.4.1 Feedstock Supply Stage

#### 6.4.1.1 Fossil-based polymers

For fossil-based polymers (PET and PP), the stage of Feedstock Supply includes the processes of crude-oil and natural gas extraction and transport, naphtha production in crude oil refineries, and its subsequent transport to downstream conversion processes (i.e. naphtha cracking). The modelling of these activities was carried out as described in Section 4.4.1.1, where the reader is referred to for further details on data sources, covered sub-processes and related environmental burdens, main modelling choices (e.g. allocation) and any adjustments performed in the implementation of the applied datasets.

For PBAT used as a copolymer in Starch-based trays, the stage of feedstock supply could not be separately modelled from downstream conversion and polymerisation processes, in the absence of disaggregated datasets related to the production of most of PBAT co-monomers (disaggregation would have only been possible for PTA). The overall, cradle-to-gate inventory modelling of PBAT production is hence entirely discussed in Section 6.4.2.3, focusing on polymer production.

#### 6.4.1.2 Recycled polymers

For recycled PET (R-PET), Feedstock Supply consists of collection of post-consumer plastic (PET) waste, and subsequent transport and sorting in specific facilities. These processes were modelled as already described in Section 4.4.5 for the Beverage Bottle case study, and refer to collection and transport of separately collected plastic waste at the municipal level (in the absence of specific data for plastic waste collection from the different sectors it may come from), as well as to sorting of mixed plastic waste in dedicated facilities. Both processes were implemented in the lifecycle model according to the Circular Footprint Formula (CFF), which is the default approach to handle recycling situations in the Product Environmental Footprint (PEF) context. According to this approach, only 50% of the burdens from collection, transport, sorting and recycling were assigned to the recycled content in trays ( $A=0.5$  for PET used in unspecified applications and for generic plastic packaging), the rest being assigned to the system providing material for recycling. Further details on the implementation of the CFF was already provided in Section 4.4.2.2.

#### 6.4.1.3 Bio-based polymers

For bio-based polymers or copolymers (Bio-PET, PLA, Thermoplastic Starch and Bio-PBS), the stage of Feedstock Supply includes cultivation of the relevant crop(s), and their subsequent transport to further processing in the same country.

Growing of Brazilian sugarcane (used as a feedstock for bio-MEG) was modelled through aggregated datasets from the GaBi database. Used in combination, these datasets depict a situation where 45% of sugarcane is manually harvested via the "slash and burn" practice, i.e. sugarcane residues (tops and leaves) are burned on standing plants before harvesting. This share was considered representative of the current situation. However, the "slash and burn" 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 phasing out of this practice (see Section 6.7.7.2). Transport of harvested sugarcane to further processing in sugarcane mills for Ethanol 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 US Maize for PLA and Bio-PBS production, and of European starch crops (maize, wheat, potatoes) used as a feedstock for Thermoplastic Starch, was modelled through available EF-compliant datasets. These are based on the approach used to develop agricultural inventories in the Agri-footprint database, relying on 5-year average yield data from FAOStat (2010-2014). Activities covered in the datasets are those of seeding and seed production, fertilizer and pesticide production and application, capital goods, as well as energy use and transport for field management practices. 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 US or EU diesel mix, depending on the country of production).

## **6.4.2 Polymer Production Stage**

The Polymer Production stage covers the activities of feedstock processing into relevant intermediates and monomer(s), the polymerisation process, compounding of polymer blends, as well as any transport among these activities and final transport of polymer granulate to the trays for food manufacturing site. The following subsections (6.4.2.1 – 6.4.2.4) describe how these activities have been modelled in the present case study.

### **6.4.2.1 Fossil-based polymers**

For conventional, fossil-based polymers (PET and PP), the whole process chain from feedstock processing to polymerisation, through the production of intermediates and monomers, was modelled by means of aggregated, gate-to-gate datasets provided by Thinkstep. Inputs includes combinations of crude oil, natural gas and naphtha, depending on the polymer. All conversion processes are assumed to take place in Europe, so that the datasets reflect EU average background conditions in terms of e.g. energy generation, material supply and transport. For both polymers, the main conversion process involved in the supply chain is steam cracking of naphtha and natural gas, delivering the monomers Ethylene and Propylene, along with Butadiene and other relevant intermediate flows (e.g. pyrolysis gas including mixtures of Benzene, Toluene and Xylenes). Other relevant conversion processes are catalytic reforming of naphtha and steam reforming of natural gas. In catalytic reforming, naphtha is processed, to produce Benzene, Toluene and Xylenes, with the latter (para-Xylene) being an intermediate in the production of Purified Terephthalic Acid (PTA, a co-monomer of PET). 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 steam cracking process of naphtha, the applied inventory is mainly based on industry data, completed, where necessary, by literature data. 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 co-products. 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 4.4.1.1), 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 generate through natural gas reforming. The final polymerisation process of PET and PP is mainly modelled based on industry data from internationally adopted production processes, integrated with literature data when needed. No allocation is performed for this stage, being PET and PP the only outputs of the respective production process.



### 6.4.2.2 Recycled polymers

The production of recycled PET granulate suitable for food trays manufacturing, out of sorted, post-consumer PET waste, was modelled based on the *ecoinvent* dataset “[CH] polyethylene terephthalate production, granulate, bottle grade, recycled”. The dataset has been developed based on data from two Swiss recycling facilities, and represents the burdens associated with the processing of sorted, pure-coloured waste PET bales into bottle-grade PET flakes, 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 assumed to 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, although is not totally clear from the dataset documentation whether this step is 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 EF datasets. The small amount of recovered HDPE (0.122 kg/kg R-PET) was assumed to directly replace virgin HDPE (1:1 substitution) according to the hierarchy for the handling of multifunctionality adopted in the PEF context and outlined in the Method<sup>93</sup>. Finally, a few adjustments had to be performed to improve reliability of LCIA results for the Ozone Depletion impact category<sup>94</sup>.

According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the recycling process<sup>95</sup> were allocated to the recycled content in PET trays ( $A = 0.5$  for PET used in unspecified applications and for generic plastic packaging). However, the recycled content carries 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 life cycle providing the recycled material). Since the  $Q_s/Q_p$  factor is equal to 1 for (food-grade) PET granules from the solid state polymerisation (SSP) process (being their quality necessarily comparable to that of virgin granules), the allocated share of virgin PET production burdens is equal to 50% ( $A \times Q_s/Q_p = 0.5 \times 1 = 0.5$ ). Virgin polymer production burdens were modelled as described above (Section 6.4.2.1) for conventional, fossil-based PET, and in Section 6.4.1.1 for the respective Feedstock Supply.

### 6.4.2.3 Bio-based polymers

#### 6.4.2.3.1 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 dataset provided by Thinkstep. The dataset covers the steps of sugarcane processing to 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 are accounted as well in the dataset, similarly to fossil-based PET. Combustion of bagasse from sugarcane processing generates surplus energy (electricity and heat) which were assumed to directly replace

<sup>93</sup> In principle, the Circular Footprint Formula (CFF) should be applied also to this minor recovered waste stream (as described further in this section for recycled PET). However, for the purpose of simplicity, direct substitution was applied, with no relevant effects on the results.

<sup>94</sup> 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 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] Market for chemical, organic), i.e. Acetic Acid, Methanol, Urea, Vinyl Acetate, Ethylene Dichloride, and Formaldehyde. The respective shares were then equally subdivided among remaining chemicals.

<sup>95</sup> Including both mechanical recycling and the subsequent solid state polymerisation process.

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. 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.

#### 6.4.2.3.2 Polylactic Acid (PLA)

For PLA production from US Maize, an aggregated, gate-to-gate inventory provided by Thinkstep was applied. The inventory includes the processes of maize wet milling for starch production, enzymatic starch hydrolysis to glucose, fermentation to lactic acid, oligomerization to lactide monomer, and final polymerisation to PLA. For the maize wet milling process, economic allocation is applied to the different co-products, while for downstream conversion processes no allocation is required. The overall inventory is mainly based on industry data from internationally adopted production processes, and is completed, where necessary, by literature data. It is noted that the dataset reflects European background conditions for energy and material inputs, despite the scenario assumes PLA production taking place in the US and subsequent transport of polymer resin to Europe. This is acknowledged as a limitation of this study.

#### 6.4.2.3.3 Starch/PBAT blend (starch-based)

Thermoplastic starch copolymer 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 methacrylate<sup>96</sup>. Moreover, in the absence of specific data on the used amount of each additive, the total share was equally split among the three mentioned substances (i.e. 8.33% each).

Starch production via wet milling of the different starch crops (maize, wheat and potatoes) was modelled based on life cycle inventory data reported in the Agri-footprint database (v 4.0), combined with background EF-compliant datasets for individual inputs and outputs. In these inventories, allocation of process burdens to the different co-products (e.g. from corn wet milling) is based on the respective economic value, consistently with the fully vertically aggregated EF datasets available for starch production via wet milling (which are developed by the same data provider). Maize starch-related 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.

Glycerol production was approximated with data related to the production of glycerine, taken from an aggregated, cradle-to-gate EF dataset. Conversely, the inventory of glucose production was based on the *ecoinvent* dataset "[RER] Glucose production", which refers to pure glucose produced via enzymatic hydrolysis of maize starch. However, in the implementation in the model, maize starch was replaced with the EU-average mix of starches from European starch crops (consistently with the starch used for the main polymer). 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 (for steam). For Glycidyl Methacrylate, an average inventory related to the production of an unspecified organic

<sup>96</sup> Based on information provided in the documentation of the GaBi dataset related to Thermoplastic Starch production.

chemical was applied (i.e. the *ecoinvent* dataset [*GLO*] *market for chemical, organic*)<sup>97</sup>,  
in the absence of more specific data on the relevant substance.

Compounding of starch and additives to form TPS was assumed to take place in the same  
facility where the final copolymer is produced by blending TPS and PBAT granulates. This  
preliminary compounding activity was not modelled separately, since the respective  
burdens are taken into account 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 synthesis (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*  
database. Inputs of PET precursors were replaced with the three PBAT co-monomers,  
considering the specific consumption reported in Schrijvers et al. (2014). These are  
based on a 90% synthesis efficiency, and are equal to 0.41 kg for Butanediol, 0.37 kg for  
adipic acid, and 0.33 kg for PTA (all expressed per kg of PBAT). Cradle-to-gate  
production inventories for these co-monomers were derived from aggregated GaBi  
datasets, with Butanediol and Adipic Acid referring to German background conditions, in  
the absence of representative data for EU conditions. Background energy inputs were  
combined with relevant EF datasets, while for material inputs and outputs the original  
*ecoinvent* datasets were kept. Infrastructure processes related to chemical factory  
construction were removed, to improve reliability of LCIA results in the Ozone Depletion  
impact category. Transport of PBAT co-monomers to the polymerisation plant was also  
accounted in the dataset, considering the default transport scenario defined in the EF  
context (and the present method) for transferring of goods from suppliers to  
factories/users located in the EU. 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 inventory of the final compounding step (including blending of raw materials, cutting,  
drying and cooling) was based on the *ecoinvent* dataset “[*RER*] Polyester-complexed  
starch biopolymer production”, which is based on calculations and extrapolations from  
highly aggregated background data from the environmental product declaration of a  
starch-based polymer (MaterBi). Compared to the original dataset, inputs of naphtha and  
natural gas were removed, as referring to the supply of raw materials and process  
energy associated with the production of fossil-based co-polyester, which is here  
represented by PBAT and was modelled separately. Similarly, the default input of maize  
starch was replaced with the average mix of starches from relevant European crops  
considered in this study. Quantities of starch, PBAT, and starch additives were set or  
adjusted so as to reflect the respective share in the final polymer blend (i.e. 30% starch,  
10% starch 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 and waste flows reported in the inventory were  
combined with EF background datasets for the modelling of the respective burdens.  
Finally, infrastructure processes related to chemical factory construction were removed,  
to improve reliability of LCIA results in the Ozone Depletion impact category.

Transport of Starch, PBAT granulate and starch additives to the compounding facility was  
also accounted in the polymer production stage. The modelling was based on the default  
transport scenario specified above for transferring of goods from suppliers to  
factories/users located in Europe, which assumes covering 130 km by lorry (> 32 t, Euro  
4), 240 km by train, and 270 km by barge.

<sup>97</sup> 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 remaining chemicals.

#### 6.4.2.3.4 Bio-based PBS

The production of fully bio-based PBS out of US corn was modelled based on an aggregated, cradle-to-gate dataset provided by Thinkstep. The dataset covers the steps of both Succinic Acid production through the enzymatic hydrolysis of wet-milled corn starch to glucose, followed by fermentation and filtration, as well as the subsequent esterification and poly-condensation of succinic acid in the presence of 1,4-Butanediol to produce polybutyl succinate. The production of 1,4-Butanediol through reduction of parts of the intermediate succinic acid is also included. For the corn wet milling process, economic allocation is applied to the different co-products, while for downstream conversion processes no other allocation seems to be required. The overall inventory is mainly based on literature data, and proxy data for a similar process are used to model the last steps of esterification and poly-condensation (limited to the respective energy consumption). It must be noted that the dataset represent a process that may undergo further improvement and optimisation in the coming years. Moreover, the dataset reflects European background conditions for energy and material inputs, despite the scenario assumes PLA Bio-PBS production taking place in the US. This is acknowledged as a limitation of this study.

Being totally aggregated (cradle-to-gate), the original dataset does not allow to separate the contribution of Feedstock Supply from downstream production burdens and impacts. Therefore, for modelling purposes, the burdens of maize cultivation in the US were artificially removed from the dataset, considering a specific consumption equal to 2.78 kg of maize grain/kg PBS (as reported in the dataset documentation) and modelling this discounted production by means of the originally applied GaBi dataset for US maize grain supply (cultivation, transport and drying).

#### **6.4.2.4 Transport of polymer granulate to the article production site**

Modelling of transport of polymer granulate from the polymerisation or compounding plant (inside or outside the EU), to the trays manufacturing site in Europe, is based on the default transport scenario (distances and vehicle types) specified in the PEF context (and in this method) for the route supplier-to-factory. In the case of polymers produced in Europe (i.e. all the polymers and copolymers investigated in this case study except for PLA and Bio-PBS -produced in USA- and for the imported share of fossil-based PET and PP), the following routes were thus considered:

- (i) 130 km by lorry (total weight >32 t; Euro 4);
- (ii) 240 km by train (average freight); and
- (iii) 270 km by ship (barge).

For the imported share of fossil-based PET and PP (21% and 10%, respectively), a transoceanic ship transport was considered as the main transport route. The corresponding overall 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)<sup>98</sup>. Countries contributing to at least 90% of the overall import were considered in the calculation, for an overall distance equal to 10,796 km for fossil-based PET, and to 8,609 km for fossil-based PP (see Tables 6.4 and 6.5). Oceanic transport was complemented with road transport to and from the harbour in both the exporting and importing country (i.e. the EU). Road transport is made by lorry (total weight >32 t; Euro 4) along an overall default distance of 1000 km.

A similar transport scenario was also assumed for PLA and Bio-PBS (entirely imported from the US), considering an estimated oceanic distance of 6,000 km (along the route New York – Rotterdam). Also in this case, a 1000 km road transport to and from the harbour in both the importing and exporting country was assumed (lorry >32 t; Euro 4).

<sup>98</sup> Available at: <https://www.searates.com/services/distances-time/>

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

**Table 6.4.** Calculation of the overall average sea distance for imports of fossil-based PET to Europe.

Exporting country	Import <sup>(1)</sup> (%)	Import (% cum.)	Distance <sup>(2)</sup> (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>

<sup>(1)</sup> Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

<sup>(2)</sup> 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 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.

1 **Table 6.5.** Calculation of the overall average sea distance for imports of fossil-based PP to Europe.

Exporting country	Import <sup>(1)</sup> (%)	Import (% cum.)	Distance <sup>(2)</sup> (km)	Weighted distance (km)
SAUDI ARABIA	41.31	41.31	8767.28	3622.14
KOREA, REPUBLIC OF (SOUTH KOREA)	13.52	54.84	16702.31	2258.83
INDIA	7.15	61.99	10267.28	734.31
RUSSIAN FEDERATION (RUSSIA)	6.75	68.74	5574.44	376.15
BRAZIL	6.63	75.37	10107.97	669.98
ISRAEL (GAZA and JERICHO- >1994)	4.86	80.23	2987.48	145.18
SOUTH AFRICA (incl. NA - >1989)	4.26	84.49	12160.92	518.45
EGYPT	3.20	87.69	3212.14	102.91
UNITED STATES	2.99	90.68	6061.90	181.22
Other countries	9.32	100	-	-
Overall weighted distance				8,609.18

2 <sup>(1)</sup> Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a).  
3 Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext  
4 data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries  
5 during a given year has been recalculated as the sum of imports from individual countries (to remove any  
6 discrepancies with the rounded total reported in raw Comext data).

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

### 12 **6.4.3 Article Production Stage**

13 Regardless of the feedstock used, manufacturing of trays for food typically takes place  
14 via thermoforming of plastic granulate. In this process, the plastic granulate is firstly  
15 plasticized in and extruded into plastic sheets of the intended thickness (passing through  
16 a die and a series of rolls). Plastic sheets are then heated into an oven to their pliable  
17 temperature and pressed into the desired shape by means of a mould.

18 The burdens of the overall conversion process were modelled through the aggregated, EF  
19 dataset "[EU-28+EFTA] Thermoforming, Plastic thermoforming, production mix, at plant,  
20 25% loss, 2.5 MJ electricity, 0.5 MJ thermal energy". The latter covers direct emissions,  
21 energy use, ancillary products and internal recycling activities related to the process, and  
22 accounts for a 97.1% conversion efficiency (i.e. 2.9% process losses, despite the value  
23 reported in the dataset name). For PET, PP, R-PET and Bio-PET trays, process losses  
24 (e.g. flawed trays) were assumed to be entirely recycled in external facilities via re-  
25 granulation into new polymer pellets, ultimately replacing virgin granules of the same  
26 material. Hence, recycled PET granules were assumed to replace virgin, fossil-based PET  
27 granules (being the estimated share of bio-based PET very low, i.e. 4%). Similarly,  
28 recycled PP granules replaced virgin, fossil-based PP granules, since no bio-based PP was  
29 reported to be available on the market at the time of developing this analysis (EUBP,  
30 2019). The recycling process and the resulting virgin material substitution was modelled  
31 based on the same data as End of Life recycling of sorted, post-consumer PET and PP  
32 trays, in the absence of more specific data for recycling of pre-consumer, industrial  
33 scraps. As for PLA, starch-based and Bio-PBS trays, process losses were assumed to be  
34 incinerated, as recycling of these polymers is not yet established (especially for polymer

blends). For further detail on the modelling of recycling and avoided virgin material production (including the implementation of the Circular Footprint Formula), as well as of material incineration, the reader is referred to Section 6.4.5 on End of Life modelling.

Transport of trays from the manufacturing site to food production/processing and packing facilities was also included in the Article Production Stage, and modelled according to the default transport scenario specified in the EF context (and the present method) for transferring of goods from suppliers to factory/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.

#### 6.4.4 Distribution Stage

The transport of the trays from the food packing site to the final user was modelled based on the default transport scenario specified in the PEF context and in this method for the route *factory* → *retail* → *final client*. The following routes were thus considered:

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

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

#### 6.4.5 End of Life Stage

This section describes the modelling of the End of Life stage of trays for food. In particular, Section 6.4.5.1 focuses on the definition of the EU-average End of Life scenario, which is considered as base case for the calculation of the potential impacts of the compared LCA scenarios. The remaining sections (6.4.5.2-6.4.5.7) address the modelling of waste collection and transport, and of the different End of Life options explored in the study. Finally, Section 6.4.5.8 provides case-specific details on the estimate of the potential contribution of trays for food to macro-plastic formation at End of Life (including product littering) and micro-plastic generation throughout the supply chain.

##### 6.4.5.1 End of Life scenario

The same EU-average End of Life scenario was considered for PET trays, regardless of the type of feedstock used for polymer production (fossil resources, plastic waste or biomass), as this does not affect the viable End of Life options for trays made of this material, nor those currently applied to them. The scenario includes 37% mechanical recycling, 33% incineration, and 30% landfilling. For PP trays, the same options are currently applied, and were included in the EU-average scenario with the following rates: 50% mechanical recycling, 26% incineration, and 24% landfilling.

The recycling rates were estimated based on the collection rates for recycling of PTTs (pots, tubes and trays) in the EU (2014) as reported by Deloitte (2017). Collection rates separately reported for PTTs generated as household waste and commercial & industrial waste<sup>99</sup> were combined based on the relative share of these two waste flows during the same year, leading to the overall collection rates specified above. Being these estimates based on more representative data for trays, their application was preferred to the default recycling rate (R2 value) reported in Annex C of the PEF method for generic plastic packaging (and to material-specific values reported for PET and PP).

<sup>99</sup> Collection rates for recycling of PTTs generated as household waste are estimated to be 31% for PET PTTs and 56% for PP ones. For PTTs in commercial & industrial waste, the collection rate is equal to 25% in the case of PET, and 42% in the case of PP.

For incineration and landfilling, EU-average rates for total plastic packaging waste were estimated and applied, in the absence of specific data for trays. The estimate was based on statistics on plastic packaging waste management in Europe for the years 2014-2016 (Eurostat, 2019b). Assuming that all the packaging waste sent to recovery operations other than recycling is incinerated, it was possible to estimate an incineration rate equal to 31%. Similarly, assuming that all the generated waste that is not recovered is landfilled, a landfilling rate of 28% could be estimated. In relative terms, 53% of plastic packaging waste that is not recycled is incinerated, while 47% is landfilled. The share of non-recycled PET and PP trays (63% and 50% respectively) was thus assumed to be routed to incineration and landfilling according to these proportions, i.e. 33% or 26% to incineration, and 30% or 24% landfilling (depending on the polymer).

For trays made of biodegradable polymers (PLA, Starch-based and Bio-PBS), currently viable End of Life options include, beyond incineration and landfilling, also biological treatment alternatives, i.e. composting and anaerobic digestion<sup>100</sup> (except for Bio-PBS, which demonstrated very low degradability under anaerobic conditions). Mechanical recycling was instead not considered a viable option for trays made of these materials, as they are primarily intended as biodegradable alternatives allowing separate collection of (food-contaminated) packaging with organic waste, potentially facilitating its management at End of Life. Moreover, even if separately collected with conventional plastic waste, they would not be sorted out for recycling with current sorting equipment, and flotation in hot caustic baths (normally used for plastic separation and cleaning during recycling) would likely initiate or further promote material degradation, making them unavailable for actual recycling.

For these biodegradable alternatives, the EU-average End of Life scenario was defined assuming that trays are separately collected with organic waste for biological treatment with the same efficiency as conventional (non-biodegradable) PET trays collected for recycling (i.e. 37%, as discussed above). This share was then split between composting (33.3%) and anaerobic digestion (3.7%), considering that over 90% of separately collected organic waste in the EU is processed into compost (ECN, 2019), the rest being handled through anaerobic digestion processes. For Bio-PBS trays, only composting was considered (with a 37% share), as discussed above. Finally, non-separately collected trays were assumed to be routed to incineration and landfilling according to the same proportions estimated above for conventional (PET) trays, i.e. 33% incineration and 30% landfilling.

#### **6.4.5.2 Modelling of waste collection and transport**

Collection and transport of (separately-collected) waste trays for mechanical recycling, incineration or landfilling 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. Similarly, relevant data to model separate collection and transport of biodegradable trays to composting or anaerobic digestion (as part of municipal organic waste), were derived from the same source. These data refer to a region with a well-developed waste management scheme in northern Italy, and can be considered representative of several regions in Europe where good levels of separate collection are achieved, with the implementation of kerbside collection systems. While this may not still be a common practice across all Europe, the approximation is considered reasonable, due to the typically moderate contribution of waste collection and transport to the overall End of Life (and lifecycle) impacts (Rigamonti et al., 2014). More information on the main assumptions and modelling details of the considered collection pathways is available in

<sup>100</sup> Degradation of biodegradable plastics under anaerobic digestion conditions is typically low, and a further aerobic (post-composting) step is normally needed to achieve suitable degradation levels. In addition, current composting standards for plastic materials and products provide only partial requirements for anaerobic digestion, i.e. in terms of mineralisation level to be achieved, while other relevant aspects (such as disintegration and quality of output material) are not addressed. However, anaerobic digestion was included in the EU-average End of Life scenario since biodegradable plastic articles collected with organic waste follow this route when it is applied as an option for such a waste stream.



Section 4.4.5.2 and, for biodegradable packaging collected with organic waste, in Section 5.4.5.2. Note that when EF incineration and landfilling datasets were applied, no transport of collected waste was separately modelled, since the respective burdens are already accounted for in such datasets. In this case, only collection was modelled, according to the approach described above.

#### **6.4.5.3 Modelling of sorting and recycling**

Before recycling, bales of separately collected plastic waste containing post-consumer trays are firstly 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 stream by colour may be performed, directly at sorting facilities or also before recovery at recycling plants.

The sorting process was modelled based on the average life cycle inventory developed by Franklin Associates (2018) for mixed plastic waste sorting in the United States (relying on data collected from different dual-stream and single-stream sorting facilities in the Country). For implementation in the model, the reported input/output data were complemented with background EF datasets representative of EU-average conditions, as better detailed in Section 4.4.5.3 (Table 4.8). A 100% sorting efficiency was assumed for the waste trays, which entirely consists of one of the targeted material for recycling (any attached lids or labels were excluded from the system boundary in this study). Moreover, no burdens from the treatment of any impurities sorted out as rejects were assigned to waste trays, to avoid falsely “punishing” them with burdens from unrelated product waste (and mostly coming from incorrect citizen behaviour). Therefore, the modelled sorting process only accounted for the material and energy inputs required to carry out this waste treatment activity.

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

As for PP (trays) recycling, no specific dataset is available in the EF context. Hence, a new dataset was developed based on foreground inventory data available again in Franklin Associates (2018), combined with EF background datasets for energy and material supply under EU-average conditions<sup>101</sup>. The inventory is based on mass-weighted average values of data collected from several recycling facilities in the United States. The overall recycling efficiency is equal to 85.5%, with removed contaminants being sent to incineration along with process waste (consistently with the typical fate of plastic recycling residues, and the assumption performed in the PET recycling process).

Recycled polymer granulate was assumed to replace virgin granulate of the same material, whose primary production burdens were credited to the system. For polymers having both a fossil-based and a bio-based alternative available on the market as of today (i.e. PET) 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

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<sup>101</sup> An aggregated EF-compliant dataset is also available for PP recycling in the US (based on an older version of the same inventory data source, i.e. Franklin Associates, 2011). However, since this could not be adjusted to reflect EU background conditions (due to its aggregated nature) and most recent data are available, a new disaggregated dataset was developed.

origin)<sup>102</sup>. To account for the lower overall quality of recycled polymers compared to the replaced virgin material, a substitution ratio equal to 0.9 was considered for PET and PP, according to the default values specified in the PEF context for such materials when used in packaging applications. To model the burdens of avoided primary production, the same datasets (or combination of datasets) used for the modelling of upstream production of the relevant polymer were applied (as described in Sections 6.4.1.1 and 7.4.1.1 for the related feedstock). This was made for consistency reasons, and avoiding possible distortions by applying different datasets from other sources. In the case of PET, this implies that avoided production of bottle-grade resins is modelled, despite the output from the recycling process is not. As a consequence the benefits associated with PET recycling are slightly overestimated<sup>103</sup>.

According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the sorting and recycling processes were allocated to the system ( $A = 0.5$  for PET and PP used in unspecified applications and for generic plastic packaging). Similarly, only 50% of the benefits from avoided virgin material production were assigned to the system itself.

#### 6.4.5.4 Modelling of incineration

For conventional fossil-based polymers (i.e. PET and PP) aggregated, material-specific incineration datasets are available from the EF database, and were applied to model the treatment of trays made of these materials in a municipal waste incineration plant. Similarly, for bio-based PET and PLA trays, and for TPS used as a copolymer in starch-based trays, partially aggregated, material-specific inventories from the GaBi database were applied (no EF datasets are available for these polymers).

All the selected datasets are based on a waste incineration model, which has been described more in detail in Section 4.4.5.4. 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 general approach to handle energy recovery situations adopted in the EF context and in the Method (Report I), the product system generating the waste material sent to incineration (i.e. the trays for food life cycle, in this case) takes the full burdens from the incineration process. However, it is credited for 100% of the benefits from avoided production of conventional energy (electricity and heat) replaced by energy recovered from waste. While in EF incineration datasets these credits are already accounted for in the aggregated inventory, for GaBi datasets they were added to the main process inventory. Therefore, the EU residual electricity grid mix (as modelled in the EF 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 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 includes 42.4% natural

<sup>102</sup> The share of bio-based PET consumed in Europe was estimated based on the total apparent PET consumption calculated from Prodcom data for the year 2015 (Eurostat, 2019c; 5323 kt), and the total bio-based PET consumption estimated for the same year in Spekrijse et al. (2019; 214.2 kt) 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).

<sup>103</sup> Note that no datasets for non-bottle-grade PET (amorphous) production are available in the GaBi database. An EF-compliant dataset is available for amorphous PET, but its application provided distorted results (especially in the Resource Use – minerals and metals impact category) and was thus excluded. The application of datasets from other sources (e.g. *ecoinvent*) was not considered, for consistency reasons and to avoid potentially larger distortions.

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. Also, thermal energy from waste (11%) was excluded, as according to the approach described above for the handling of energy recovery situations, 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 starch-based trays and for Bio-PBS, no ready-to-use incineration datasets were available. A disaggregated, material-specific inventory was thus developed, based on the most recent version of the calculation tool developed by Doka (2009a) for the modelling of material incineration within municipal solid waste incineration plants. The model operates similarly to the one used for the development of EF incineration datasets, 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 6.6 for an overview of the considered composition values). The tool also allows to adjust other technological parameters to the relevant geography or scope, including energy efficiencies, the share of alternative NO<sub>x</sub> control technologies, and a few other specific parameters. In this case 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, and applied in the modelling. These efficiencies account for the share of waste routed to incineration plants operating without any energy recovery (estimated to be 9% for municipal waste)<sup>104</sup>, while considering that plants with energy recovery operates with an average gross electricity efficiency equal to 15.1%, and a gross thermal efficiency equal to 35% (CEWEP, 2012)<sup>105</sup>. Credits associated with recovered energy were calculated as described above (i.e. electricity from the EU-average 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, background EF datasets were applied.

#### 6.4.5.5 Modelling of landfilling

Landfilling of trays made of non-biodegradable polymers (i.e. virgin and recycled fossil-based PET and fossil-based PP) was modelled based on a common, aggregated, EF dataset representing disposal of plastic waste in a managed, municipal solid waste landfill (*[EU-28+EFTA] Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific sites)*). The underlying inventory is material-specific, but refers to the average chemical composition and degradability values of generic plastic waste, rather than to those of the specific polymer. 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 landfill modelling) is similar for all non-biodegradable (conventional) polymers. 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 emissions to the environment over a 100 year time horizon. Emissions occurring beyond 100 years are not accounted for in the model. Landfill gas generation is calculated based on the organic carbon content in the waste and the respective degradation rate assumed over 100 years (not reported), while, for simplification reasons, an average landfill gas composition for the stable methane

<sup>104</sup> Calculated as average share of municipal waste incinerated without energy recovery over the years 2015-2017 (Eurostat, 2019d).

<sup>105</sup> 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).

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 rate of landfill gas, energy efficiencies of engines, collection rate of leachate and respective treatment efficiencies). Further details on how these parameters were set to reflect the EU-average situation are provided in Section 4.4.5.5.

The same dataset described above for landfilling of generic plastic waste was also applied as proxy for landfilling of trays made of “drop-in” non-biodegradable, bio-based polymers i.e. partially bio-based PET. Compared to the original dataset, emissions of CO<sub>2</sub> and CH<sub>4</sub> to air have been converted from fossil to biogenic emissions, to reflect the bio-based origin of carbon in such polymer. 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 amounts of fossil CO<sub>2</sub> and CH<sub>4</sub> reported as released to air were converted to biogenic emissions (disregarding at the same time the only partial bio-based origin of carbon in Bio-PET trays). This approximation is considered acceptable, as reported CO<sub>2</sub> and CH<sub>4</sub> emissions are modest, and overall amounting to only 1.4% of the carbon content in the landfilled plastic material, despite they also include the contribution of emissions from background processes. This can be partly explained by the generally low degradation (and mineralisation) of conventional, non-biodegradable polymers in landfill, which is typically assumed to be in the range of 1% over 100 years (Doka, 2009b). On the other hand, it is acknowledged that this approach is in favour of this partially bio-based polymer.

In line with the time horizon applied for landfill emission modelling in the selected dataset, (biogenic) carbon in the landfilled polymer that is not degraded (mineralised) after 100 years from deposition was considered to be never released from the landfill body. As a base case, this missed release of atmospheric carbon taken up during biomass growth is not reflected in Climate Change LCIA results for bio-based PET trays, since characterisation factors for biogenic CO<sub>2</sub> flows have been set to zero (fully conforming to provisions in the PEF method). However, to show the relevance of this choice on the overall results, the Climate Change impact of Bio-PET trays was also calculated by acknowledging the effects of non-released biogenic carbon. For this purpose, a specific CO<sub>2</sub> uptake was modelled in the inventory, expressing the net amount of CO<sub>2</sub> taken up during biomass growth and not released (as CO<sub>2</sub> or CH<sub>4</sub>) during polymer degradation in landfill. The uptake was calculated based on the biogenic carbon content in the polymer, and assuming a mineralisation rate over 100 years equal to 1% (Doka, 2009b). While the mineralisation rate assumed in the dataset is not reported explicitly, it seems to be in line with this assumption (according to the figures reported above regarding the inventoried carbon emissions). Considering a biogenic carbon content in Bio-PET equal to 12.5% (i.e. 20% of a total carbon content of 62.5%), a carbon uptake equal to 0.454 kg CO<sub>2</sub>/kg Bio-PET was calculated. This uptake was then characterised with a factor of -1 kg CO<sub>2</sub> eq. per kg CO<sub>2</sub>-C not degraded.

For trays made of biodegradable polymers (i.e. PLA, starch-based and Bio-PBS), a material-specific landfilling inventory was built, based on the calculation tool developed by Doka (2009b) for the modelling of waste disposal into sanitary landfills<sup>106</sup>. The tool allows to calculate material-specific landfilling inventories accounting for the specific chemical composition and other relevant chemo-physical properties of the landfilled waste (Table 6.6), as well to adjust a number of relevant site- and technology-specific parameters to the reference geography and to the corresponding average landfilling technology. In addition, the model applies specific transfer coefficients to calculate the distribution of decomposition products originating from elements in the waste composition between landfill gas and leachate, and their ultimate emissions 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

<sup>106</sup> For starch-based trays, separate inventories were developed for the two copolymers (i.e. TPS and PBAT).

1 afterwards due to temporary storage in the landfill body (e.g. metals liberated from the  
2 waste matrix and then re-precipitated in solid form). Delayed emissions only include  
3 waterborne emissions with non-collected leachate (which are inventoried separately),  
4 while air emissions with landfill gas entirely take place over the first 100 years from  
5 deposition.

6 In line with the time horizon applied for landfill emission modelling in this tool and in the  
7 earlier reported dataset, (biogenic) carbon in the landfilled polymers that is not degraded  
8 after 100 years from deposition was considered to be never released from the landfill  
9 body. Therefore, also in this case, to calculate a variant of Climate Change LCIA results  
10 that acknowledge the effects of non-released atmospheric carbon, a specific CO<sub>2</sub> uptake  
11 was modelled in the inventory, expressing the net amount of CO<sub>2</sub> taken up during  
12 biomass growth and not released (as CO<sub>2</sub> or CH<sub>4</sub>) during polymer degradation in landfill.  
13 Considering the biogenic carbon content reported for each polymer in Table 6.6, the  
14 following carbon uptakes were calculated, based on the mineralisation rates reported  
15 above (i.e. 1% for both PLA and Bio-PBS trays, and 11% for TPS used in starch-based  
16 trays): 1.80 kg CO<sub>2</sub>/kg PLA, 2.03 kg CO<sub>2</sub>/kg Bio-PBS, and 1.46 kg CO<sub>2</sub>/kg TPS (i.e. 0.585  
17 kg CO<sub>2</sub>/kg starch-based polymer).

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**Table 6.6.** Elemental composition and lower heating value of PLA, TPS, PBAT and Bio-PBS considered for End of Life modelling.

Element	(%)	Element	(%)
<b>PLA <sup>(1)</sup></b>			
TS	100	H (%TS)	5.6
Water	0	O (%TS)	44.5
VS (%TS)	100	N (%TS)	0.1
Ash (%TS)	0	S (%TS)	0.3
C (%TS)	49.5	C <sub>biogenic</sub> (% total C)	100
LHV = 18.4 MJ/kg TS <sup>(2)</sup>			
<b>TPS <sup>(3)</sup></b>			
TS	100	C (%TS)	44.8
Water	0	H (%TS)	6.58
VS (%TS)	100	O (%TS)	48.6
Ash (%TS)	0	C <sub>biogenic</sub> (% total C)	100
LHV = 17.3 MJ/kg TS <sup>(2)</sup>			
<b>PBAT <sup>(4)</sup></b>			
TS	100	C (%TS)	53.7
Water	0	H (%TS)	6.47
VS (%TS)	100	O (%TS)	39.8
Ash (%TS)	0	C <sub>biogenic</sub> (% total C)	100
LHV = 21.1 MJ/kg TS <sup>(2)</sup>			
<b>Bio-PBS <sup>(4)</sup></b>			
TS	99.9	C (%TS)	55.8
Water	0.1	H (%TS)	7.0
VS (%TS)	100	O (%TS)	37.2
Ash (%TS)	0	C <sub>biogenic</sub> (% total C)	100
LHV = 22.1 MJ/kg TS <sup>(2)</sup>			

<sup>(1)</sup> Based on results from composition analysis related to Ingeo polymer available at: <https://www.natureworksllc.com/What-is-Ingeo/Where-it-Goes/Incineration>.

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

<sup>(3)</sup> Based on stoichiometry, and considering that TPS consists of 75% starch and 25% plasticisers/additives (equally split among Glycerol, Sorbitol and Glycidyl Methacrylate).

<sup>(4)</sup> Based on stoichiometry.

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. carbon to CH<sub>4</sub> and CO<sub>2</sub>) within the landfill. For PLA trays, a 1% biodegradation was assumed, according to the results from Accelerated Landfill Conditions (ALC) tests simulating approximately 100 years of deposition in a biologically active landfill (390 days at 21 °C, Kolstad et al., 2012). These showed no significant biodegradation of both amorphous and semi-crystalline PLA over such a time horizon, even at the highest humidity level applied for testing (i.e. 65%,

which should be considered in a conservative approach). Therefore, the same value of degradability assumed in the model for conventional, non-biodegradable polymers such as PET and PE was consistently applied (i.e. 1%). Similarly, for starch-based film (or better, its copolymers TPS and PBAT), results from ALC testing of Mater-Bi (NF 803 grade) at 65% humidity were considered (Vermeulen, 2007). In this case, a biodegradation of nearly 11% was observed (based on net biogas production), although the testing timeframe was lower than above (i.e. 180 vs 390 days) and the degradation curve (evolution of net biogas production) had not yet reached a plateau, but was still increasing. Hence, a higher biodegradation rate can actually be expected over a 100 year timeframe. In the absence of Accelerated Landfill Conditions (ALC) tests for PBS, a 1% biodegradation was assumed for Bio-PBS trays, in light of the negligible biodegradation of this material under anaerobic digestion conditions (see section 7.4.5.7).

Site-specific and technology-specific parameters were set so as to reflect as much as possible the average situation at the EU level, as detailed in Section 4.4.7.5. These include the rates of landfill gas capture, utilisation and flaring, as well as the mean annual precipitation and temperature. For the other parameters (e.g. energy efficiencies of gas engines, mean annual evapotranspiration, landfill height and duration of the filling phase) the default values assumed in the model were kept.

In the final datasets, 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 database<sup>107</sup>. However for energy-related flows, background EF datasets were applied.

#### **6.4.5.6 Modelling of composting and compost use-on-land**

A waste-specific life cycle inventory was developed for aerobic composting of biodegradable trays (i.e. PLA, starch-based and Bio-PBS trays), following the approach and the assumptions described in Section 5.4.5.6 (to which the reader is referred for further details beyond those specified below). The modelling was carried out according to the general recommendations specified in the Method (Report I - Section 4.4.12.4 and 4.4.12.7), while relying on process-specific burdens and element-specific transfer coefficients available in the EASETECH model (Clavreul et al., 2014) for the assumed composting technology (i.e. tunnel composting).

The elemental composition considered in the modelling for each tray material is the one already reported in Table 6.6, while the applied biodegradation rates are summarised in Table 6.7. The latter were defined as the average of (experimental) values available in the literature for fragments of plastic sheets/bars, films or similar items. Since these values were at least partly determined under real composting or laboratory conditions, they were preferred to the application of the minimum biodegradation rate required by the European standard for compostable plastic packaging (EN 13432) i.e. 90%. This has indeed to be proven under optimised laboratory testing conditions and may not be achieved in real composting plants (or it may anyway differ from values obtained from real laboratory testing available in the literature).

Consistently with the process modelled for food packaging film (Section 5.4.5.6), a first screening step was considered to be carried out before composting. At this stage, 30% of the incoming plastic trays were assumed to be removed, regardless of their specific shape or size at that point in time, and sent to incineration (modelled as described in Section 6.4.5.4).

Carbon in biodegradable trays sent to composting was assumed to mineralise according to the biodegradation rates reported in Table 6.7. The same rates were also assumed for biodegradation of volatile solids (VS) in each tray material. According to the emission factors (transfer coefficients) reported in the EASETECH model, 99.8% of mineralised carbon is converted to CO<sub>2</sub>, the rest (0.2%) being converted to CH<sub>4</sub>. The latter is then

<sup>107</sup> Exceptions are the inputs of diesel (burned in building machine) and pitch, which were replaced with suitable EF datasets, to improve reliability of LCIA results in the Ozone Depletion impact category.

1 mostly oxidised to CO<sub>2</sub> (95%), while only 5% is eventually emitted as Methane, equalling  
2 0.01% of the mineralised carbon. No waste-specific Nitrogen emissions from the  
3 composting process were modelled, being the Nitrogen content of the investigated  
4 biodegradable polymers equal to zero or negligible (e.g. for PLA).

5 The amount of residual composted material (biomass and simpler organic compounds)  
6 obtained from the biodegradation of trays was calculated as the sum of non-degraded  
7 volatile solids, as well as of ash and water<sup>108</sup> originally included in the material sent to  
8 composting. The residual material was assumed to be entirely applied, by means of  
9 tractors, on agricultural land, where biodegradation of further advance. The emissions  
10 from on-land application were estimated based on average emission factors estimated by  
11 Bruun et al. (2006), considering a time horizon of 100 years from application, and  
12 average factors across all types of temperate soils. During this period, 89.3% of the  
13 applied carbon was mineralised to Carbon Dioxide, 0.01% was released as Methane,  
14 while the remaining 10.7% was not released<sup>109</sup> (being part of more stable organic  
15 compounds).

16 The overall composting inventory was implemented in the lifecycle model according to  
17 the Circular Footprint Formula, considering it as a case of recycling with a value of the A  
18 factor equal to 0.5. Only 50% of the burdens from the composting process were thus  
19 allocated to the trays life cycle. Conversely, the burdens and benefits from on-land  
20 application of the residual composted material were entirely allocated to it.

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<sup>108</sup> It is acknowledged that the water content in the residual composted material may change compared to the original material sent to composting due to losses and integration during the process. However, the water content of the investigated biodegradable polymers is equal to zero, so that this simplifying assumption does not affect the calculated amount of residual material from composting.

<sup>109</sup> As a base case, the effects of non-released biogenic carbon are not reflected in Climate Change LCIA results for biodegradable trays made of bio-based polymers, since the characterisation factors for biogenic CO<sub>2</sub> flows have been set to zero (fully conforming to provisions in the PEF method). The consequences of this are not significant, since the share of residual, non-emitted carbon is low, equalling 0.5-5% of total carbon in the composted polymers. However, to show the relevance of this missed quantification on comparative scenario results, the Climate Change impact of PLA, starch-based and Bio-PBS trays was also calculated by acknowledging the effects of non-released biogenic carbon. For this purpose, a specific CO<sub>2</sub> uptake was modelled in the composting inventory, corresponding to the amount of biogenic carbon not released over 100 years, which is equal to 0.0654 kg CO<sub>2</sub>/kg PLA trays sent to composting, 0.0147 kg CO<sub>2</sub>/kg starch-based trays, and 0.00764 kg CO<sub>2</sub>/kg Bio-PBS trays.



**Table 6.7.** Biodegradability values considered for biodegradable trays for food (PLA, starch-based and Bio-PBS) under aerobic composting conditions.

Polymer	Biodegradation rate (%)	Source
PLA	51.8 <sup>(1)</sup>	Deconinck and De Wilde (2013) – Grinded bars; 3.2 mm thickness (88%) Hermann et al. (2011) – Film fragments; 1 mm thickness; 1 cm <sup>2</sup> area (55%) UBA (2018) – Pot fragments (13%)
Starch-based	70	UBA (2018) – TPS/PCL (Mater-Bi class Z; 90%) Deconinck and De Wilde (2013) – Starch/PCL blend (90%) Hermann et al. (2011) – different reviewed values for Mater-Bi and Starch/PCL films and bags (27% film, 60%, 75% film, 90% bags)
Bio-PBS	95 <sup>(2)</sup>	UBA (2018) Deconinck and De Wilde (2013)

<sup>(1)</sup> Calculated as the average of (experimental) results for fragments of PLA sheets/bars with 1 mm and 3.2 mm thickness and pot fragments.

<sup>(2)</sup> Calculated as the average of experimental results for Bionelle polymer.

#### 6.4.5.7 Modelling of anaerobic digestion

Anaerobic digestion of PLA and starch-based trays<sup>110</sup> and subsequent post-composting of the resulting, non-digested or partially digested, residual material were modelled according to the approach and the assumptions described in Section 5.4.5.7 (to which the reader is referred to for further details beyond those specified below). For anaerobic digestion, a waste-specific inventory was developed, based on the general recommendations specified in the Method (Report I – Section 4.4.12.5), while relying on relevant process-specific burdens and element-specific transfer coefficients available in the EASETECH model (Clavreul et al., 2014) for the assumed digestion technology (i.e. wet thermophilic digestion).

The elemental composition considered in the modelling is the one already reported in Table 6.6, while the applied biodegradation/mineralisation rates under anaerobic digestion conditions are summarised in Table 6.8. The latter were defined as the average of (experimental) values available in the literature for plastic films (except for PLA, where only one value was considered to be reliable). Since these values have been at least partly determined under real digestion or laboratory conditions, they were preferred to the application of the minimum biodegradation rate required by the European standard for compostable plastic packaging (EN 13432), i.e. 50% when testing anaerobic treatability. This has indeed to be proven under optimised laboratory testing conditions, and may not be achieved in real digestion plants (or it may anyway differ from values obtained from real laboratory testing available in the literature).

Similarly to the composting process, prior to digestion the incoming waste material was assumed to undergo a pre-treatment stage, where it is also shredded to make it suitable for further processing in the digestion plant. In this stage, 30% of biodegradable trays collected for treatment were assumed to be removed as residues, and sent to incineration (modelled as described in Section 6.4.5.4).

Carbon in biodegradable trays sent to digestion was assumed to be converted (mineralised) into biogas according to the biodegradation rates reported in Table 6.8. The shares of gasified carbon converted into Methane and Carbon Dioxide were calculated

<sup>110</sup> Bio-PBS has very low degradability under anaerobic conditions (UBA, 2018) so that anaerobic digestion was not assessed for trays made of this polymer.

based on the stoichiometry of the anaerobic degradation reaction, taking into account the proportions of C, H, O and N in the digested polymers. Carbon conversion shares are thus material-specific and range from 50% (PLA) to 53% (starch-based polymer) for Methane, and from 47% (starch-based polymer) to 50% (PLA) for Carbon Dioxide.

**Table 6.8.** Biodegradability values considered for biodegradable trays for food (PLA and starch-based) under thermophilic anaerobic digestion conditions.

Polymer	Biodegradation rate (%)	Source
PLA	25	UBA (2018) – PLA film -0.1 mm-
Starch-based	22	UBA (2018) – TPS/PCL (Mater-Bi) 20 µm film (23%) <sup>1</sup> UBA (2018) – Starch/co-polyester (Ecobras) 720 µm film (20%) <sup>2</sup>

<sup>(1)</sup> Refers to mesophilic digestion conditions (35°C), but aligns with the other reported value (which refers to thermophilic conditions and to a duration of 50 days).

<sup>(2)</sup> Refers to a duration of 50 days, which appears long for thermophilic digestion conditions.

The amount of residual, non-digested or partially digested bioplastic material in the digestate output was calculated as the sum of non-degraded volatile solids (VS), as well as of ash and water<sup>111</sup> originally present in the input material to digestion. Residual volatile solids were in turn calculated by assuming a ratio between degraded VS and degraded carbon equal to 1.89 (according to the value applied to organic waste in the EASETECH model).

Aerobic post-composting of such residual material from digestion was modelled considering an open (windrow) composting facility, along with the typical average carbon and VS biodegradation (mineralisation) rate achieved for generic organic waste in this type of facilities (i.e. 76.4% according to the considered EASETECH dataset). This rate was consistently applied to residual organic material derived from digestion of all types of biodegradable polymers considered for trays manufacturing, regardless of the level of biodegradation achieved in the first anaerobic digestion step. Based on transfer coefficients (emission factors) reported in the mentioned windrow composting dataset, 97.76% of mineralised carbon was assumed to be converted to CO<sub>2</sub>, and the remaining 2.24% to CH<sub>4</sub>.

The residual organic material from post-composting of digested trays was assumed to be applied on agricultural land as soil amendment. On-land application was modelled according to the same approach and assumptions briefly described in Section 6.4.5.6 for residual material derived from direct composting of biodegradable trays (and more extensively detailed in Section 5.4.5.6 for the food packaging film case study).

#### **6.4.5.8 Contribution to macro- and micro-plastics generation**

The contribution to macro- and micro-plastics generation of the analysed food trays scenarios was estimated according to the framework outlined in Annex B of the Method (Report I), and summarised in Section 4.4.5.6 of this report in terms of underlying quantification methods and approaches. In this section, the case-specific parameters considered to apply the latter to the assessed food trays LCA scenarios are reported.

##### **6.4.5.8.1 PLP method**

Regarding the estimate of the loss and release of macro-plastics at the End of Life stage (from product littering and waste mismanagement), the product-specific parameters reported in Table 6.9 were considered to apply the first level of the framework ("PLP

<sup>111</sup> It is acknowledged that the water content in the digestate output may change compared to the original material sent to digestion due to losses and integration during the process. However, the water content of the investigated biodegradable polymers (PLA and Bio-PBS) is equal or very close to zero, so that this simplifying assumption does not affect the calculated amount of residual material from digestion.

*method*”) to trays for food (derived from Peano et al., 2020). Note that such parameters are defined regardless of the type of feedstock or material used for trays manufacturing. Similarly, all trays for food scenarios contribute to macro-plastics loss and release to the same extent, as the chosen feedstock or material do not affect the probability of the article to be littered.

**Table 6.9.** Case-specific parameters of the *PLP method* applied to trays for food LCA scenarios.

Parameter <sup>(1)</sup>	Value
Littering rate (LR <sub>lit</sub> ) (%)	2
Release rate to ocean (Rel <sub>ocean</sub> ) (%)	25
Release rate to the terrestrial environment (Rel <sub>terenv</sub> ) (%)	75

<sup>(1)</sup> For details on the meaning of each parameter, refer to Annex B of the Method (Report I).

No case-specific parameters are required to calculate the supply-chain loss and release of micro-plastics to ocean and to the terrestrial environment by applying the *PLP method*. Indeed, the contribution of the considered micro-plastics sources (pellet loss and tire abrasion) depends on the magnitude of the relevant life cycle processes (i.e. polymer production and road transport) in the specific food trays scenario (which in turns depend on the reference flow<sup>112</sup> of each scenario). However, the quantification is made by means of default parameters that are not affected by the type of product, polymer or feedstock source (as reported in Annex B of the Method (Report I)).

#### 6.4.5.8.2 Expanded PLP method

The parameters required to calculate the micro-plastics loss and release from marine coatings and road markings are not case-specific, and are reported in Annex B of the Method (Report I). Similar to the other considered micro-plastics sources, the contribution of these additional sources depends on the intensity of relevant (transport) processes in the specific product inventory, by means of the above-mentioned case-unspecific parameters.

As for the calculation of the additional contribution of plastic waste exported for recycling to the loss and release of macro-plastics from waste mismanagement, the case-unspecific parameters described in Annex B of the Method (Report I) were applied, in combination with the EU-average recycling rates estimated for each type of trays for food (Section 6.4.5.1). For the sake of clarity, the applied recycling rates are summarised again in Table 6.10. Note that some of the considered bio-based scenarios are considered to be composted instead of following a recycling path.

**Table 6.10.** EU-average recycling rates considered for each trays for food scenario to estimate the amount of waste plastic trays exported to non-EU countries for recycling.

Polymer	Recycling rate
PET	37%
PP	50%
PLA	0%
Starch-based	
R-PET	37%
Bio-PBS	0%
Bio-PET	37%

<sup>112</sup> The reference flow is the mass of tray material required to fulfil the functional unit.

#### 6.4.5.8.3 Sensitivity analysis 1: alternative calculation of the total plastic loss along the supply-chain

This sensitivity analysis was not applied to the food trays case study.

#### 6.4.5.8.4 Sensitivity analysis 2: alternative calculation of macro-plastics release to ocean

The EU-average marine litter rate (MLR, %) for trays for food and the resulting release to ocean (kg) were estimated based on beach litter observations and apparent consumption data at the EU level, according to the procedure reported in Annex B of the Method (Report I). For calculation purposes, overall values for the period 2012-2016 were considered for both quantities. Moreover, the following product categories were considered in the respective data sources, reported as well below:

- Beach litter observations (Beach litter database; Addamo et al., 2018; Hanke et al., 2020): “*Cutlery, trays, straws, stirrers*” (B18), as no specific category was available for food trays.
- EU Apparent consumption (PRODCOM database; Eurostat, 2019c): “*Tableware and kitchenware of plastic*”, since no specific category for food trays was available.

Table 6.11 summarises the values considered in the calculation, along with the obtained estimates of the beach litter rate and of the resulting (up-scaled) marine litter rate.

**Table 6.11.** Estimate of the beach litter rate and of the resulting up-scaled marine litter rate for trays for food.

Total predicted beach litter <sup>(1)</sup> (kg/year)	Total apparent EU consumption <sup>(2)</sup> (kg/year)	Beach Litter rate (%)	Marine Litter rate (%)
2.58E+05 <sup>(3)</sup>	9.80E+08	0.0263	0.526

<sup>(1)</sup> Along the entire length of EU coast. Estimated based on observed beach litter data over 100 m of coastline (from Addamo et al., 2018 and Hanke et al., 2020).

<sup>(2)</sup> From PRODCOM (Eurostat, 2019c).

<sup>(3)</sup> Assumed average weight = 4.68g per unit.

### 6.4.6 Calculation of iLUC impacts

As a base case, the contribution of Indirect Land Use Change (iLUC) to the potential Climate Change impact of bio-based polymers was calculated according to the approach outlined in Section 4.4.16.4 of the Method (Report I).

In order to apply (recalculated) iLUC GHG emission factors from the EU 2015/1513 Directive (EC, 2015), the specific land demand for the production of relevant crops (m<sup>2</sup>\*year / kg crop) was calculated first. The calculation was based on the aggregated amount of arable and agriculture land occupation flows reported, for the relevant country (e.g. Brazil for sugarcane), in the dataset used to model the production of the specific crop. If the geography of such flows was not specified, all the arable and agricultural land occupation flows reported in the dataset were aggregated. These estimates were checked against values of land demand calculated based on 5-years average crop yields from FAOSTAT (FAO, 2019), which were found to be generally aligned with the former (absolute variation between 3% and 20%). Hence, the values estimated based on land occupation flows were applied, for consistency with the overall LCI modelling of scenarios.

Specific land demand for crop production was then converted into a demand per functional unit, based on the specific consumption of crop for polymer production (kg crop/kg polymer, consistently with the values applied in the LCI modelling) and the amount of polymer needed to fulfil the functional unit (reference flow) in the relevant scenario. The iLUC contribution to the Climate Change impact was finally calculated by applying to the latter the recalculated GHG emission factors from the Directive. The

- 1 described calculation steps to estimate the iLUC contribution to the potential Climate
- 2 Change impact are summarised in Table 6.12.
- 3

1 **Table 6.12.** Calculation of the iLUC contribution to the potential Climate Change impact of trays for food LCA scenarios relying on bio-based polymers.

Scenario / Polymer	Feedstock	Land demand for crop production <sup>(1)</sup> [m <sup>2</sup> *y/kg <sub>crop</sub> ]	Crop demand for polymer production [kg <sub>crop</sub> /kg <sub>polymer</sub> ]	Polymer demand per F.U. [kg <sub>polymer</sub> /FU]	iLUC GHG emission factor [kg CO <sub>2</sub> eq./ (m <sup>2</sup> *y)]	iLUC climate change impact [kg CO <sub>2</sub> eq./FU]
S4 – Bio-PET	Sugarcane (BR)	0.117 (0.135)	4.27	170	0.176	14.9
S5 – PLA	Maize (US)	1.06 (0.939)	3.08	62.8	0.0612	12.6
S6 – Starch-based	Maize (EU)	1.34 (1.37)	0.134	173	0.0612	1.89
	Wheat (EU)	1.52 (1.75)	0.121		0.0612	1.95
	Potatoes (EU)	0.376 (0.301)	0.140		0.0612	0.557
	<i>Total</i>	-	-		-	4.40
S7 – Bio-PBS	Maize (US)	1.06 (0.939)	2.78	95.9	0.0612	17.4

(2) Based on arable and agriculture land occupation exchanges reported in the dataset applied to the modelling of the specific crop. Values in parenthesis refer to land demand calculated based on crop yield data from FAOSTAT (5-years average), and are reported as a reference.

## 6.5 Life Cycle Impact Assessment results

The characterised potential impacts of the examined scenarios (with and without the iLUC contribution to Climate Change) are reported in Figures 6.8 to 6.10. Note that these impacts are the result of the performance prioritised in the FU (i.e. oxygen barrier performance) for all cases but PP (see section 6.2). These also show the breakdown of contributions from the main lifecycle stages, which include:

- i) *Feedstock Supply*, i.e. oil/natural gas extraction, transport and possible refining, transport of naphtha from refinery to downstream users; collection, transport, and sorting of post-consumer plastic waste (recycled polymers); or crop cultivation and transport to further processing (bio-based polymers);
- ii) *Polymer Production*, i.e. all gate-to-gate activities from feedstock processing to monomer/intermediate production and polymerisation, including any transport among these, as well as transport of polymer granulate to the trays for food manufacturing site;
- iii) *Article Production*, i.e. conversion of the polymer into trays for food by thermoforming, and subsequent transport to downstream users (i.e. producers of packaged food);
- iv) *Distribution*, i.e. transport of trays for food from the food packing site to the final user, in the form of a defined packaging item (e.g. lidded or wrapped trays); and
- v) *End of Life*, i.e. waste trays collection, transport and treatment or disposal, as well as any avoided processes from downstream displacement of virgin materials and energy.

Figure 6.11 shows the contribution of each sub-category to the Climate Change impact (i.e. from fossil, biogenic and land use/land transformation GHG emissions), while the overall characterised, normalised and weighted impact assessment results are available in Annex B.3 (Tables B.3.8 to B.3.10). The latter are not reported in this section to keep readability of the document, despite this is not in alignment with the position suggested in the LCA report template provided in the Method (Report I).

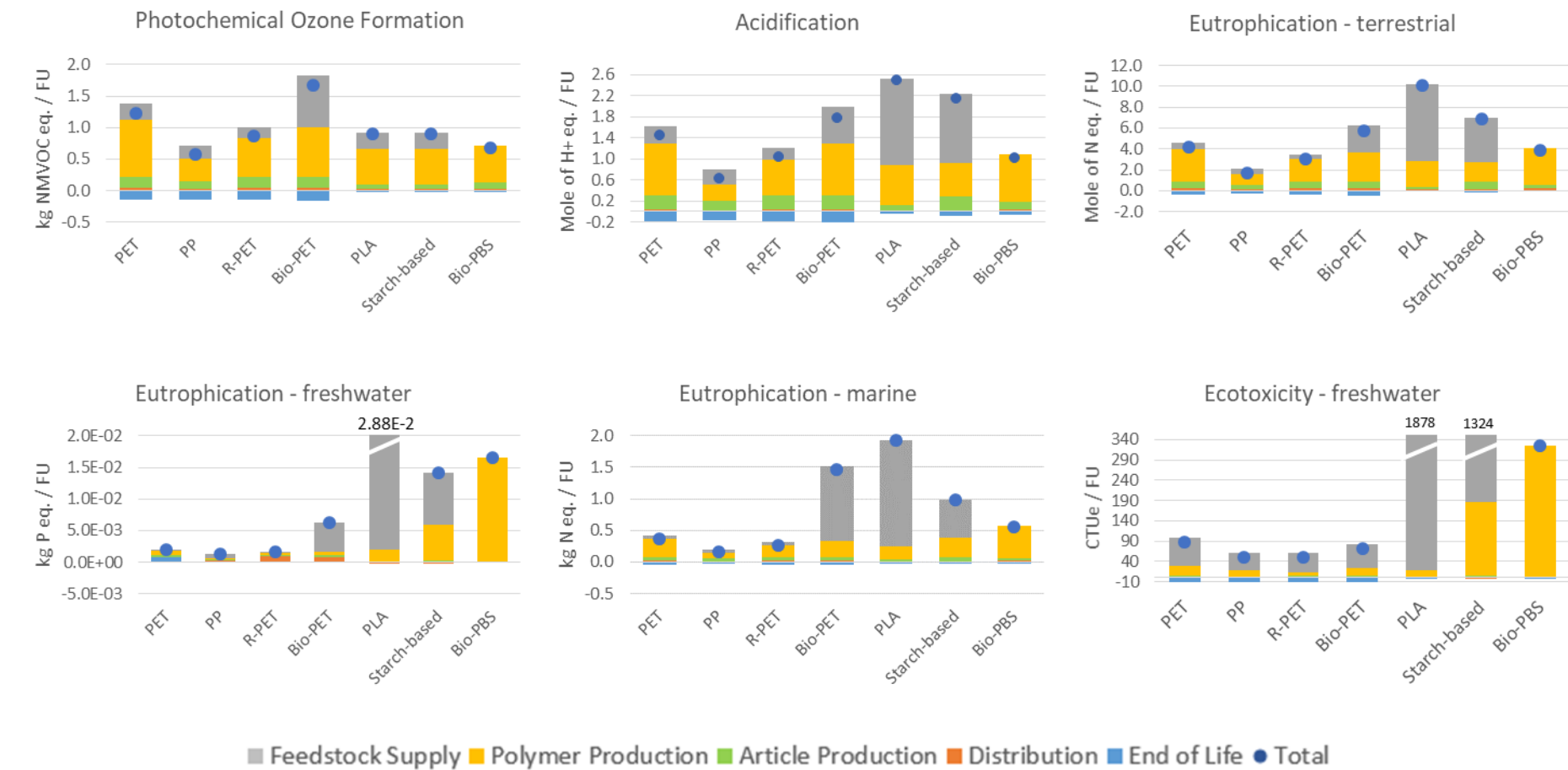
Note that scenario impacts presented in Figures 6.8 to 6.10 refer to the EU-average End of Life scenario (as described in Section 6.4.5.1), and represent the net impacts from waste management activities, resulting from the balance between real burdens and benefits (if any). Potential impacts calculated assuming 100% of post-consumer trays being routed to each viable End of Life option are presented in Figure 6.12 to 6.14. We warn that this analysis should not be interpreted as a direct LCA comparison of alternative End of Life management options for trays for food, because a product-based perspective is considered, and allocations among consecutive product life cycles are applied to the recycling pathway. Conversely, in a typical waste management LCA that compares alternative End of Life options (i.e. with a functional unit of 1 tonne of waste managed), the recycling pathway would be assigned the full burdens and benefits it involves, as there is no need to break the system mass flows between the system recycling and the one using the recycled feedstock (i.e. system perspective; no allocation needed).

## Comparative LCA of Alternative Feedstock For Plastics Production – DRAFT FOR STAKEHOLDER CONSULTATION –Part II

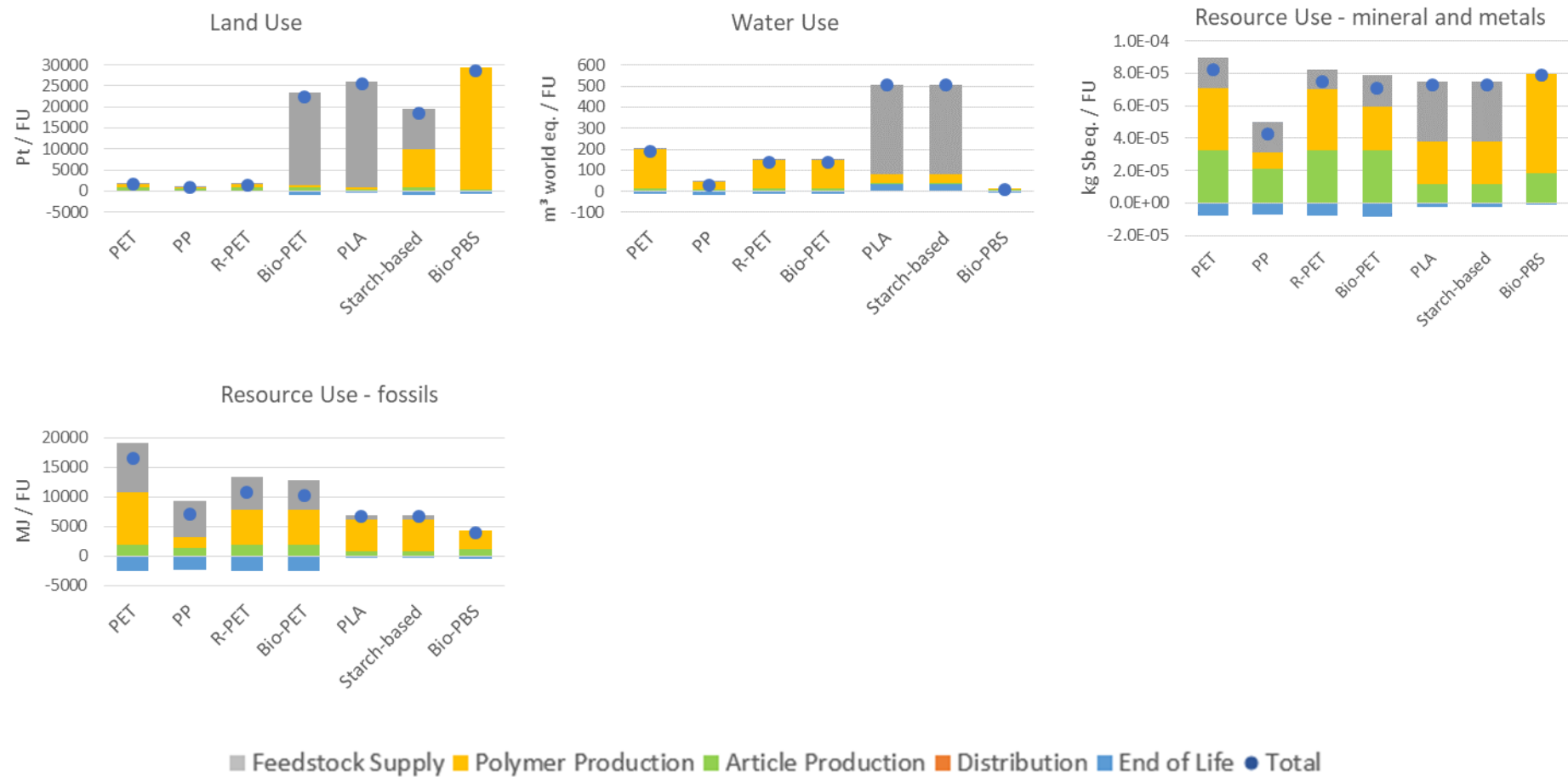


**Figure 6.8.** Potential impact of trays for food LCA scenarios for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity - non-cancer, Particulate Matter and Ionising Radiation. Climate Change impacts denoted with "C bio EoL" accounts for the contribution of biogenic carbon not released after 100 years from landfilling of bio-based trays for food. Note that the Polymer Production stage for bio-based PBS trays refers to an aggregated, cradle-to-gate dataset, which also covers the impacts from Feedstock Supply. Note also that in some impact categories a part of the results is out of scale and is curtailed.



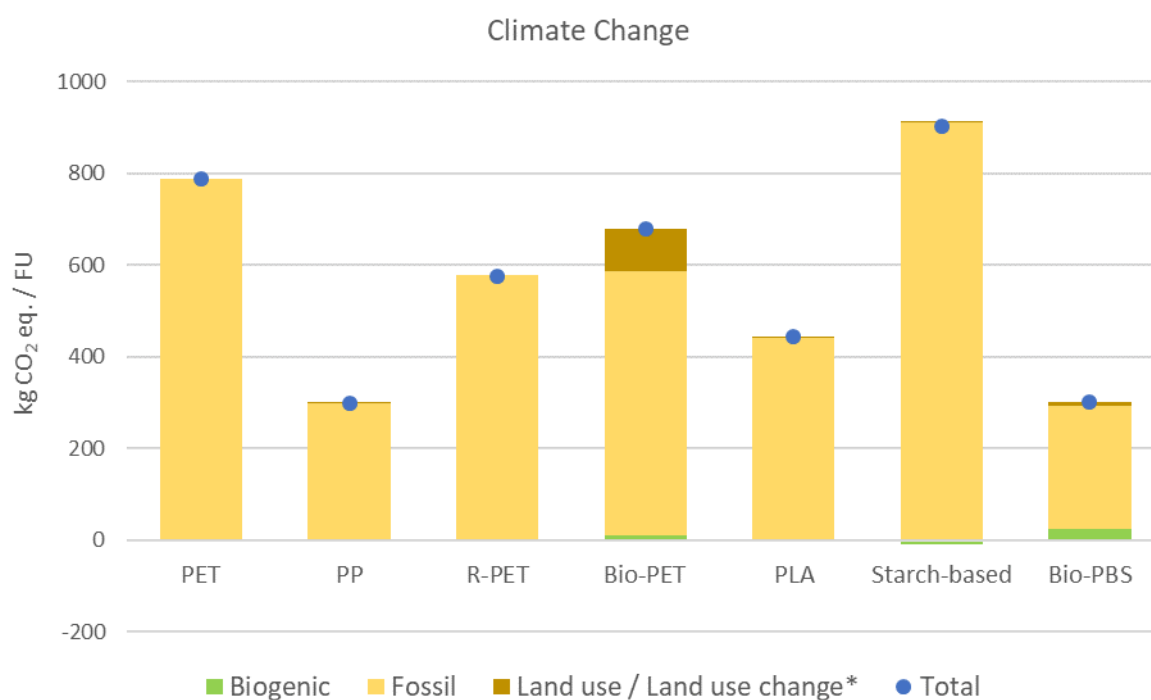


**Figure 6.9.** Potential impact of trays for food LCA scenarios for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater. Note that the Polymer Production stage for bio-based PBS trays refers to an aggregated, cradle-to-gate dataset, which also covers the impacts from Feedstock Supply. Note also that in some impact categories a part of the results is out of scale and is curtailed.



**Figure 6.10.** Potential impact of trays for food LCA scenarios for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils. Note that the Polymer Production stage for bio-based PBS trays refers to an aggregated, cradle-to-gate dataset, which also covers the impacts from Feedstock Supply.

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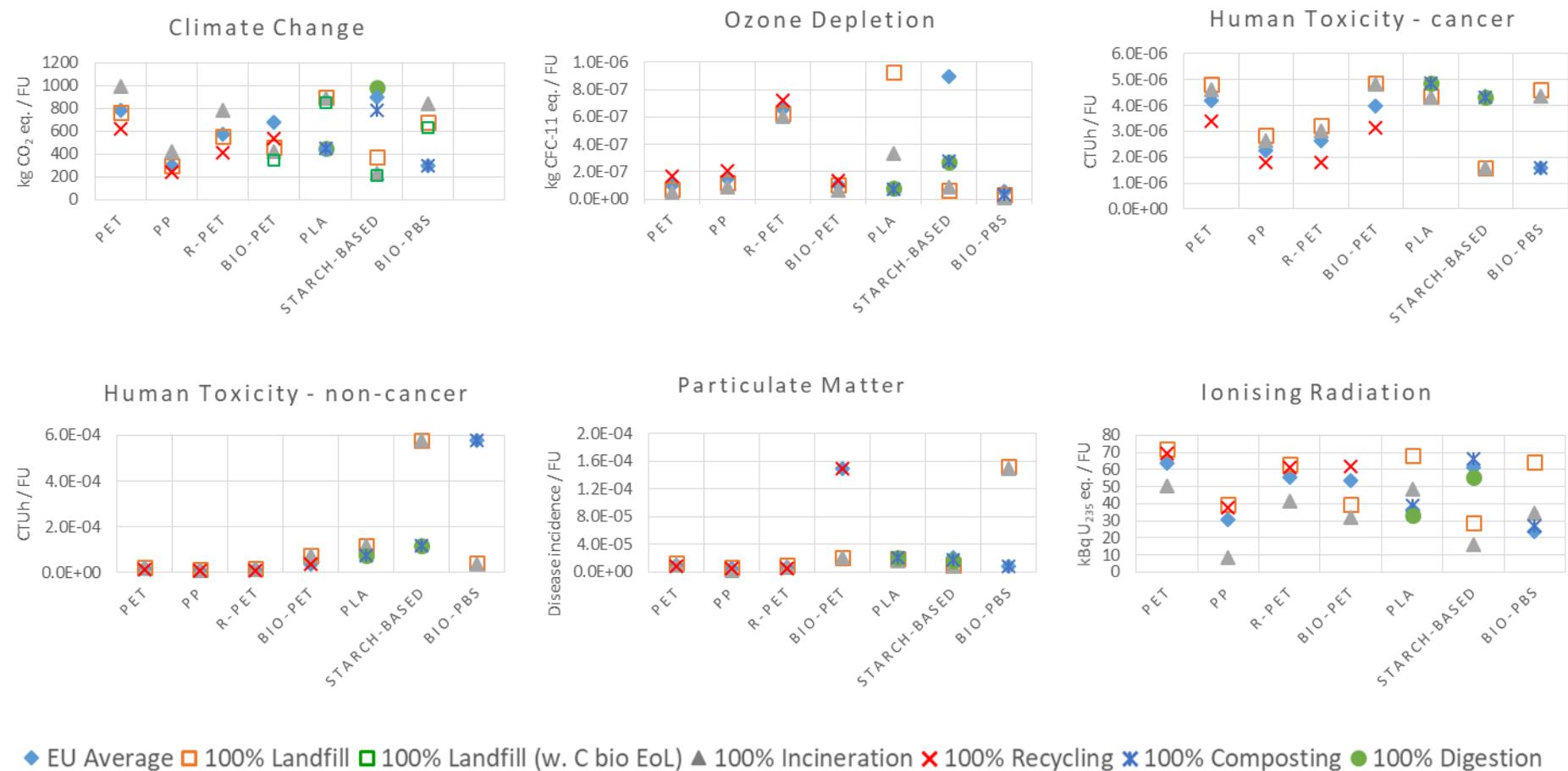
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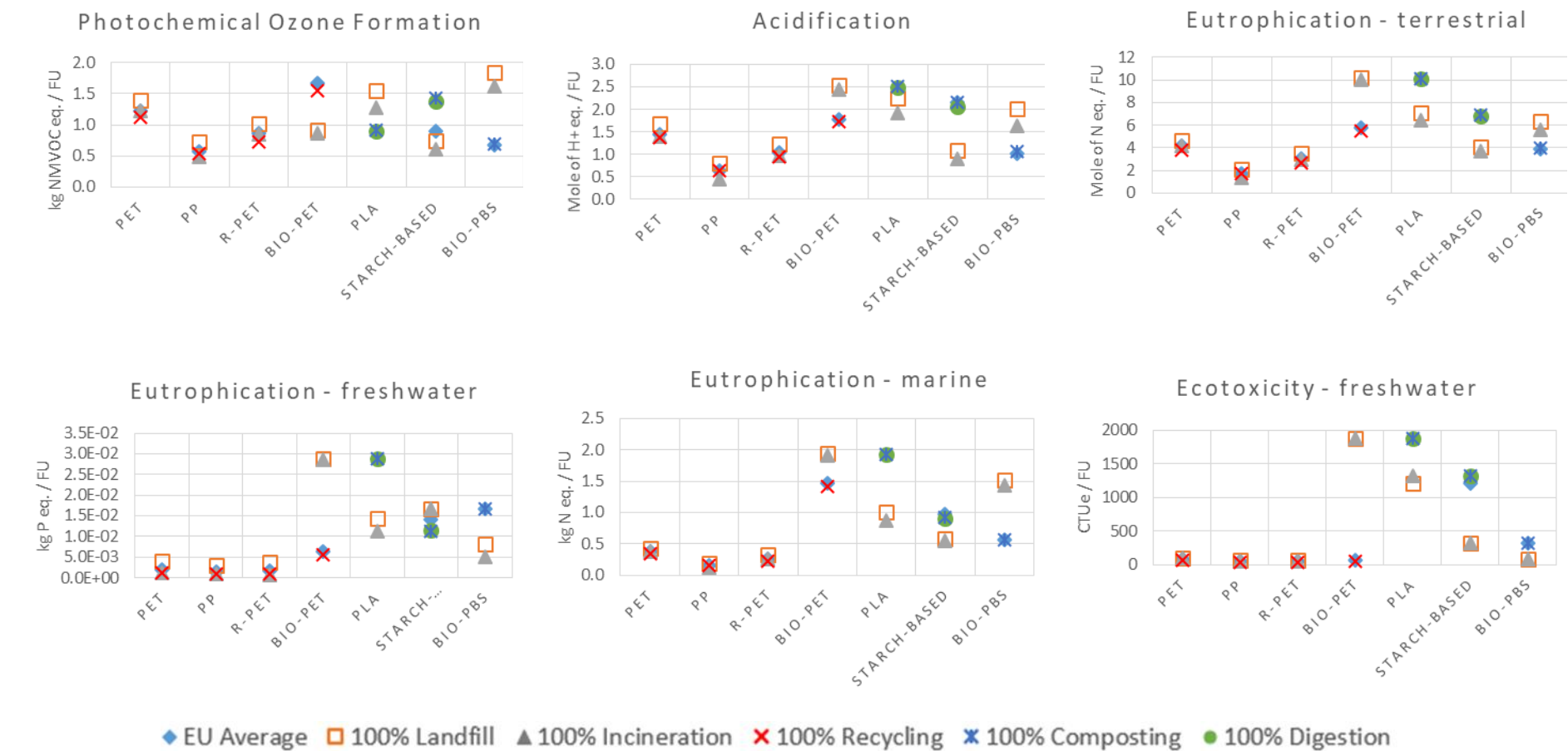
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**Figure 6.11.** Contribution of fossil, biogenic, and land use/land transformation GHG emissions to the total Climate Change impact of trays for food LCA scenarios (note that the contribution of iLUC to land transformation emissions is excluded).

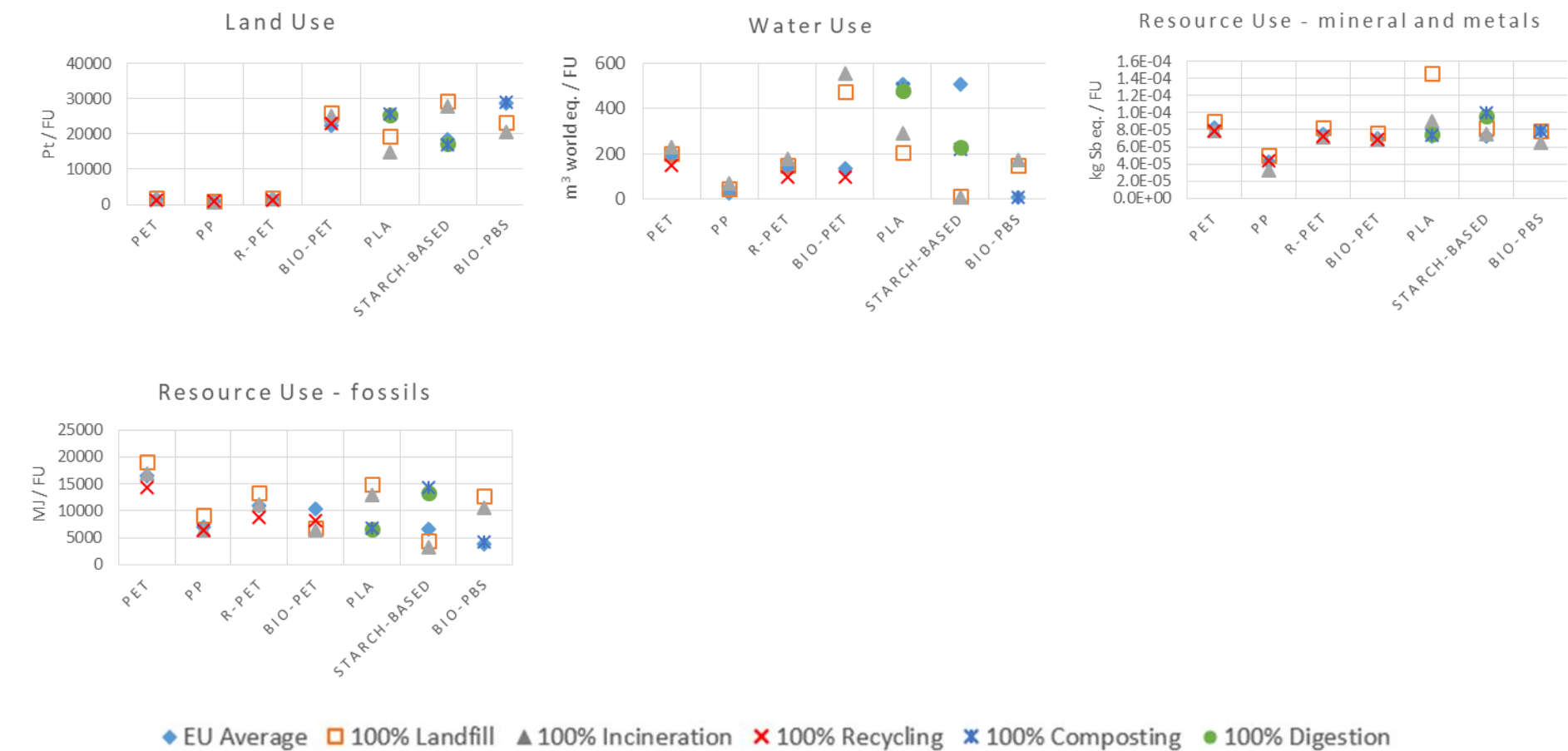
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**Figure 6.12** Potential impact of trays for food LCA scenarios for different End of Life options, for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity – non-cancer, Particulate Matter and Ionising Radiation. The Climate Change impact of 100% landfilling scenarios denoted with “C bio EoL” accounts for the contribution of biogenic carbon not released after 100 years from landfill deposition of bio-based trays for food.



**Figure 6.13** Potential impact of trays for food LCA scenarios for different End of Life options, for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater.



**Figure 6.14** Potential impact of trays for food LCA scenarios for different End of Life options, for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils.

## 6.6 Additional Environmental Information

This section presents additional environmental impacts or aspects going beyond the default impact categories considered in the present method, which reflect official categories adopted in the PEF context. Additional environmental issues addressed include the impact from indirect Land Use Change (iLUC) on Climate Change, potential Biodiversity impacts, as well as the contribution of trays for food to macro-plastic formation at End of Life (including product littering) and micro-plastic generation throughout the supply chain.

### 6.6.1 iLUC impact

Table 6.13 presents the estimated contribution of GHG emissions from iLUC induced by bio-based feedstock supply to the potential Climate Change impact of bio-based trays for food.

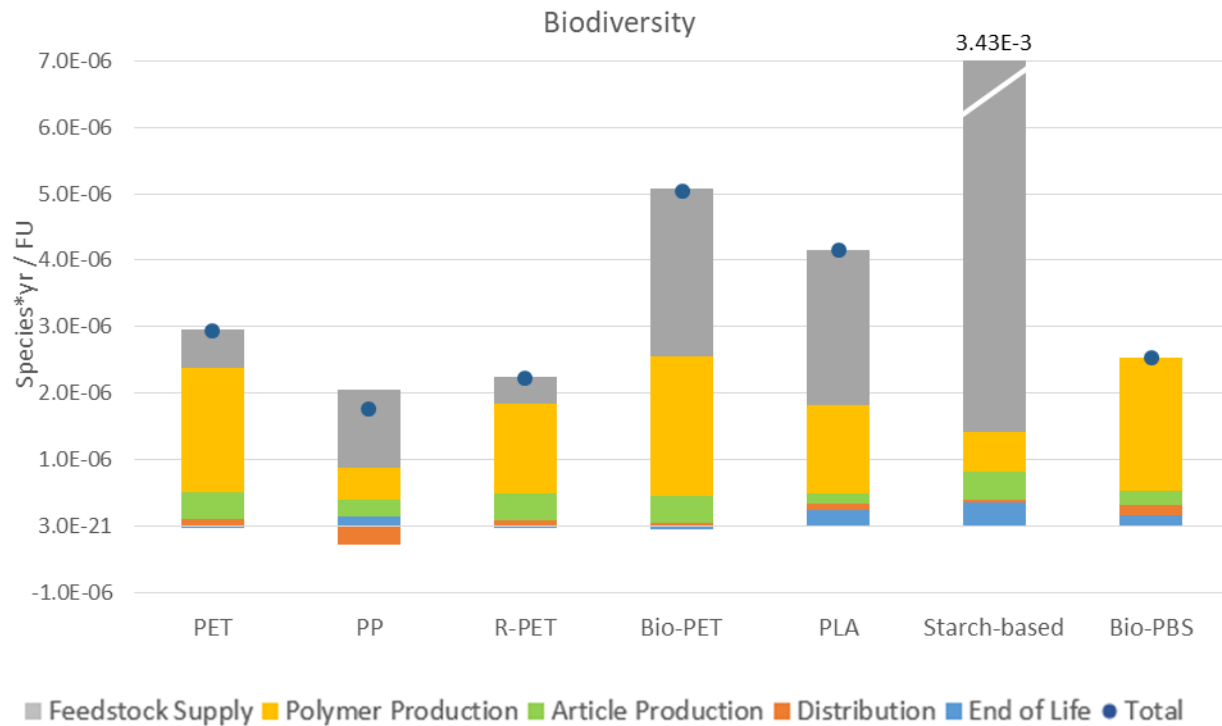
**Table 6.13.** iLUC contribution to the potential Climate Change impact of trays for food LCA scenarios.

Scenario / Polymer	iLUC Climate Change impact [kg CO <sub>2</sub> eq/FU]	Total Climate Change impact (incl. iLUC) <sup>(1)</sup> [kg CO <sub>2</sub> eq/FU]
S1 – PET	-	(783)
S2 – PP	-	(299)
S3 – R-PET	-	(578)
S4 – Bio-PET	14.9	665(679)
S5 – PLA	12.6	456 (444)
S6 – Starch-based	4.40	907 (902)
S7 – Bio-PBS	17.4	319 (301)

<sup>(1)</sup> Values in parenthesis refer to the Climate Change impact without the iLUC contribution.

### 6.6.2 Biodiversity impacts

Estimated potential Biodiversity impacts of the investigated scenarios are presented in Figure 6.15. It is reminded that the assessment is performed through an endpoint-level indicator relying on impact assessment methods that for some of the underlying midpoint impact drivers differ from those recommended in the present method.

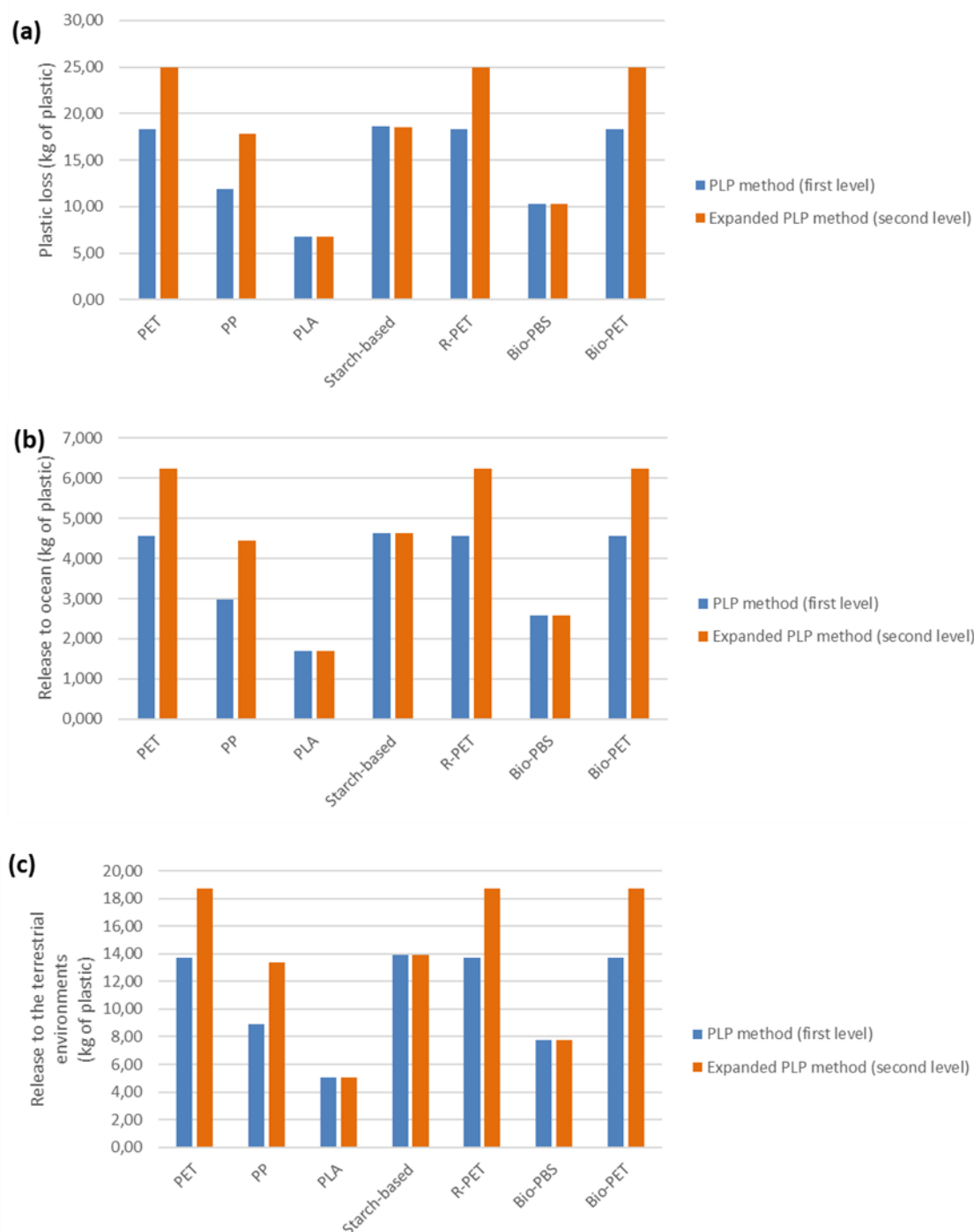


**Figure 6.15.** Potential biodiversity impact of trays for food LCA scenarios, expressed as potential loss of species per year.

**6.6.3 Macro- and micro-plastic generation**

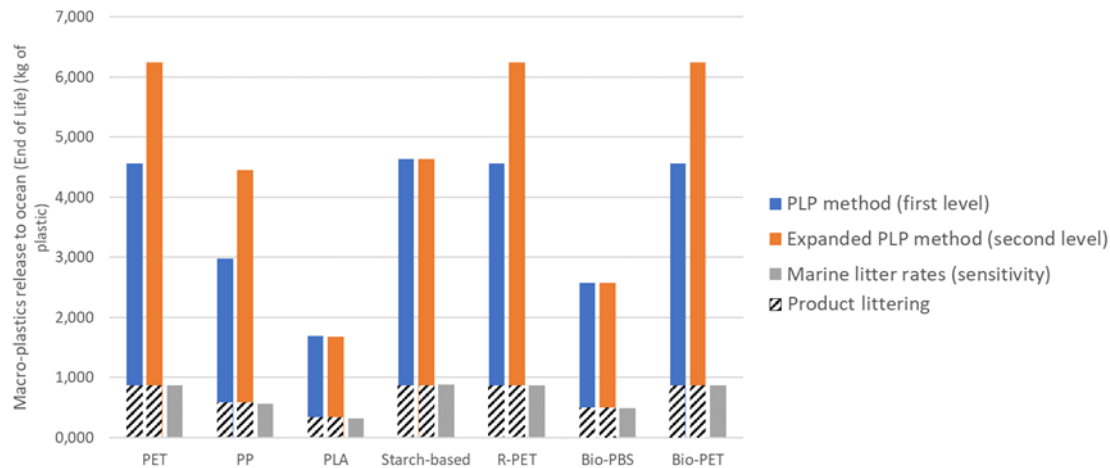
Figure 6.16 shows the total plastic loss, release to ocean and release to the terrestrial environment estimated for the assessed trays for food scenarios, considering the whole product life cycle. Estimates obtained by applying all the two different explored approaches are presented, including the total loss and release to ocean/terrestrial environment calculated through both the PLP method (first level) and the Expanded PLP method (second level).





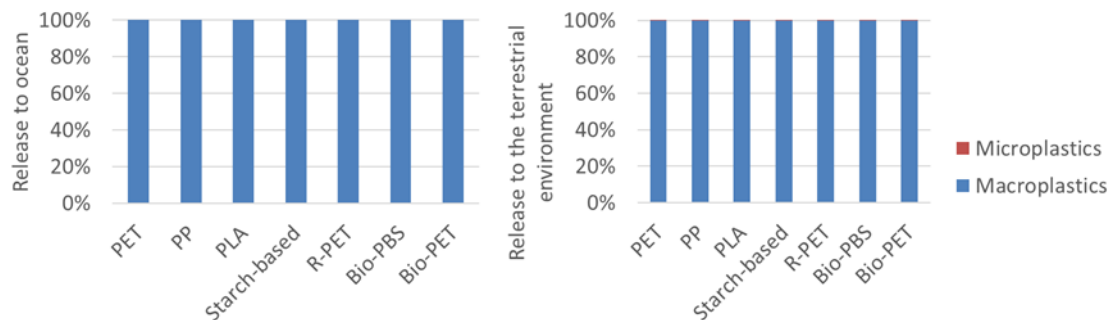
**Figure 6.16.** Total plastic loss (a), release to ocean (b) and release to the terrestrial environment (c) estimated for trays for food LCA scenarios with different approaches. The contribution of both macro- and micro-plastics is included, considering the whole product life cycle.

Regarding the sole release of macro-plastics to ocean at End of Life (from product littering and waste mismanagement), Figure 6.17 presents the contribution of each food trays scenario, estimated with the three different explored approaches: the PLP method (first level), the Expanded PLP method (second level), and the sensitivity approach based on product-specific marine litter rates from observed beach litter in the EU. Note that the latter do not differentiate the type of source (e.g. littering and waste mismanagement), but inherently takes into account all the possible sources that may ultimately contribute to macro-plastic release to ocean at the European level (still with a higher degree of uncertainty).



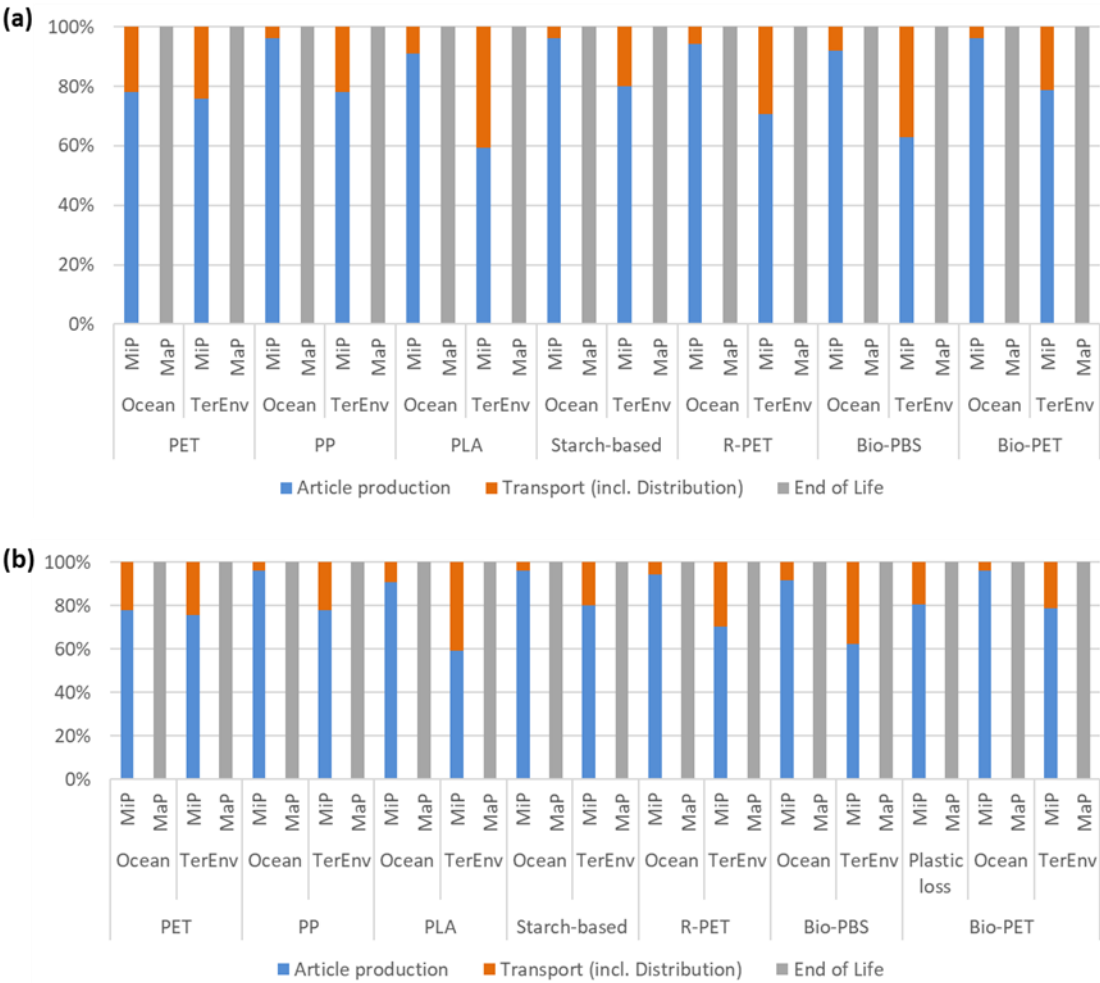
**Figure 6.17.** Macro-plastics release to ocean at End of Life, estimated with different approaches.

As for the share between the release of macro- and micro-plastics, Figure 6.18 shows as an example the relative contribution of the two categories to the total release to ocean and to the terrestrial environment estimated through the PLP method (first level). Note that the results are similar to those obtained when applying the Expanded PLP method (second level), as further discussed in Section 6.7.7.



**Figure 6.18.** Contribution of macro- and micro-plastics to the total plastic release to ocean and to the terrestrial environment estimated with the PLP method for each food trays LCA scenario.

The contribution of the different lifecycle stages to the total macro- and micro-plastics release to ocean and to the terrestrial environment of the assessed food trays scenarios has been finally evaluated. The results of this “hotspot analysis” are reported in Figure 6.19 for both the PLP method and the Expanded PLP method. Results are not shown for the other sensitivity approach as it covers only a part of the product life cycle (i.e. End of Life).



**Figure 6.19.** Contribution of life cycle stages to the total macro-plastics (MaP) and micro-plastics (MiP) release to ocean (Ocean) and to the terrestrial environment (TerEnv) estimated for trays for food LCA scenarios with (a) the PLP method (first level) and (b) the Expanded PLP method (second level).

## 6.7 Interpretation

In the interpretation of case study results, most relevant impact categories and lifecycle stages of the studied systems are firstly identified (6.7.1 and 6.7.2, respectively). Scenario impacts are then compared, considering both characterised LCIA results (6.7.3), and total normalised and weighted impacts (6.7.4). Results obtained by applying individually each viable End of Life option are separately discussed in Section 6.7.6, while the effects of iLUC and the contribution to macro- and micro-plastics generation are addressed in Sections 6.7.5 and 6.7.7, respectively. Finally, the results of the sensitivity analysis on relevant parameters and assumptions are presented (6.7.8).

Most relevant processes were not identified, since it was not possible to achieve a similar level of vertical disaggregation for foreground processes included in the life cycle models of the analysed scenarios (e.g. a higher disaggregation could be achieved for Starch-based trays, but this was not the case of other scenarios). Therefore, the identification of most relevant processes cannot be carried out consistently across all scenarios, and a more detailed investigation for specific scenarios would not be meaningful. The identification of most relevant elementary flows was also not undertaken, as this would require prior identification of most relevant processes. Note, however, that any company, organisation or any other supply chain actor applying the present method shall proceed with the identification of both most relevant processes and elementary flows

### 6.7.1 Identification of most relevant impact categories

Table 6.14 shows the most relevant impact categories identified for each scenario, based on normalised and weighted impacts, according to the approach outlined in the Method (Report I, Section 6.2.1). Relevant categories were identified as those that cumulatively contribute to at least 80% of the total normalised and weighted impact of the considered impact categories, excluding toxicity related ones (these being still based on characterisation factors implemented in the EF 2.0 impact assessment methods, and hence not yet updated based on REACH data<sup>113</sup>). 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 recycled (fossil-based) PET trays, and also for fossil-based PP trays, two impact categories are identified as the most relevant, i.e. Climate Change and Resource Use - fossils. The latter show a similar contribution in all the mentioned scenarios (39-44%), although in the case of PP trays the order of relevance is the opposite (with Resource Use – fossils being the most relevant category).

In the case of Bio-PET trays, Particulate Matter is the most relevant impact category, followed again by Climate Change, and Resource Use – fossils. As for PLA and starch-based trays, the most relevant impact categories include Climate Change, Resource Use – fossils, Water Use, Particulate Matter and, limited to PLA trays, also Acidification. Finally, for Bio-PBS trays, Climate Change, Resource Use – fossils, Land Use and Particulate Matter were found to be the most relevant impact categories (i.e. a similar ranking as starch-based trays, with the exception that Water Use is replaced by Land Use as the third most relevant category).

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<sup>113</sup> According to the latest version of the Product Environmental Footprint Category Rules Guidance (EC, 2018), toxicity-related impact indicators based on these characterisation factors shall be excluded from the procedure to identify most relevant impact categories.

**Table 6.14.** Most relevant impact categories identified for trays for food LCA scenarios and related contribution to the total normalised and weighted scenario impact.

S1 - Fossil-based PET		S2 – Fossil-based PP		S3- Recycled PET	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Climate Change	41.9%	Resource Use – fossils	44.2%	Climate Change	44.0%
Resource Use – fossils	41.8%	Climate Change	39.8%	Resource Use – fossils	39.0%
Acidification	3.2%	Acidification	3.4%	Acidification	3.3%
<b>Total</b>	<b>86.9%</b>	<b>Total</b>	<b>87.4%</b>	<b>Total</b>	<b>86.2%</b>
S4 – Bio-PET		S5 – PLA		S6 – Starch-based	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Particulate Matter	33.7%	Climate Change	32.0%	Climate Change	51.3%
		Resource Use – fossils	22.7%	Resource Use – fossils	17.9%
Climate Change	29.5%	Water Use	10.0%	Water Use	7.9%
		Particulate Matter	7.9%	Particulate Matter	6.2%
Resource Use - fossils	21.1%	Acidification	7.4%		
<b>Total</b>	<b>84.3%</b>	<b>Total</b>	<b>80.0%</b>	<b>Total</b>	<b>83.3%</b>
S7 – Bio-PBS					
<i>Impact category</i>	<i>Contrib.</i>				
Climate Change	40.6%				
Resource Use – fossils	25.0%				
Land Use	8.5%				
Particulate Matter	11.4%				
<b>Total</b>	<b>80.2%</b>				

### 6.7.2 Identification of most relevant life-cycle stages

Table 6.15 shows the most relevant lifecycle stages in the relevant impact categories identified in Section 6.7.1, and the associated contribution, quantified according to the rules detailed in the Method (Report I, Section 6.2.2). Most relevant stages include those that together contribute to at least 80% of the total characterised lifecycle impact in the specific category, and are highlighted in yellow in Table 6.15. For a more exhaustive picture, the contribution of all lifecycle stages is reported.

For fossil-based PET trays, Polymer Production is identified as the most relevant stage, along with Article Production (in Climate Change) and Feedstock Supply (in Resource Use – fossils). For fossil-based PP trays, Feedstock Supply is identified as the sole most relevant stage in Resource Use – fossils (85% of the total impact), while in Climate Change as it is preceded by Polymer Production and followed by Article Production. For

24% recycled PET trays, Polymer Production is the most relevant stage, as along with Article Production (in Climate Change) and Feedstock Supply in (Resource Use – fossils).

For bio-based PET trays, Polymer Production and Feedstock Supply are identified as the most relevant stages, in Climate Change and Resource Use – fossils. In Particulate Matter the stage of Feedstock Supply contributes, alone, to 95% of the overall impact, thus being the only relevant life-cycle stage.

For PLA trays, Feedstock Supply is the most relevant life cycle stage for Water Use (where it is responsible for 90% of the total impact) and, along with Polymer Production, also for Particulate Matter and Acidification. For Climate Change and Resource Use – fossils, Polymer Production is the most relevant stage, along with Feedstock Supply in the case of Climate Change.

The most relevant life cycle stage for starch-based trays is Feedstock Supply, except for Resource Use – fossils, where Polymer Production is the most relevant one. In Climate Change and Particulate matter, Polymer Production and (limited to Climate Change) Article Production are also part of most relevant stages, baryon Feedstock Supply. As for bio-based PBS trays, Polymer Production is the most relevant life cycle stage (covering also the contribution of Feedstock Supply), along with Article Production in Climate Change.

**Table 6.15.** Contribution of life cycle stages to the total characterised impacts of most relevant categories for trays for food LCA scenarios. Most relevant stages (i.e. those contributing to more than 80% of the total impact) are highlighted in yellow.

S1 – Fossil-based PET					
Climate Change		Resource Use - fossils		Acidification	
Life cycle stage	Contrib.	Life cycle stage	Contrib.	Life cycle stage	Contrib.
Polymer Production	74.3%	Polymer Production	53.5%	Polymer Production	68.8%
Article Production	13.9%	Feedstock Supply	49.7%	Feedstock Supply	22.7%
Feedstock Supply	8.0%	Article Production	11.3%	Article Production	18.0%
End of Life	2.8%	Distribution	0.6%	Distribution	3.0%
Distribution	0.9%	End of Life	-15.2%	End of Life	-12.5%
S2 – Fossil-based PP					
Resource Use – fossils		Climate Change		Acidification	
Life cycle stage	Contrib.	Life cycle stage	Contrib.	Life cycle stage	Contrib.
Feedstock Supply	85.3%	Polymer Production	49.3%	Polymer Production	49.7%
Polymer Production	28.5%	Feedstock Supply	24.0%	Feedstock Supply	45.9%
Article Production	17.3%	Article Production	23.7%	Article Production	27.0%
Distribution	0.8%	End of Life	1.6%	Distribution	4.3%
End of Life	-31.9%	Distribution	1.5%	End of Life	-26.9%

1

S3 – Recycled PET					
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	67.9%	Polymer Production	54.1%	Polymer Production	65.8%
Article Production	18.9%	Feedstock Supply	50.9%	Article Production	24.9%
Feedstock Supply	8.1%	Article Production	17.2%	Feedstock Supply	22.5%
End of Life	3.8%	Distribution	0.9%	Distribution	4.2%
Distribution	1.3%	End of Life	-23.1%	End of Life	-17.4%
S4 – Bio-based PET					
Particulate Matter		Climate Change		Resource Use - fossils	
<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	95.0%	Polymer Production	58.4%	Polymer Production	57.6%
Polymer Production	4.9%	Feedstock Supply	24.7%	Feedstock Supply	47.5%
Article Production	1.7%	Article Production	16.1%	Article Production	18.2%
Distribution	0.1%	Distribution	1.1%	Distribution	1.0%
End of Life	-1.7%	End of Life	-0.2%	End of Life	-24.2%
S5 – PLA					
Climate Change		Resource Use - fossils		Water 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>
Polymer Production	72.6%	Polymer Production	80.7%	Feedstock Supply	83.9%
Feedstock Supply	20.2%	Feedstock Supply	11.9%	Polymer Production	8.7%
Article Production	9.1%	Article Production	10.4%	End of Life	6.5%
Distribution	0.9%	Distribution	0.5%	Article Production	0.9%
End of Life	-2.8%	End of Life	-3.4%	Distribution	0.0%
Particulate Matter		Acidification			
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>		
Feedstock Supply	62.5%	Feedstock Supply	66.0%		
Polymer Production	34.2%	Polymer Production	30.7%		
Article Production	4.6%	Article Production	3.8%		
Distribution	0.4%	Distribution	1.0%		
End of Life	-1.7%	End of Life	-1.5%		

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S6- Starch-based					
Climate Change		Resource Use - fossils		Water Use	
Life cycle stage	Contrib.	Life cycle stage	Contrib.	Life cycle stage	Contrib.
Feedstock Supply	57%	Polymer Production	80.7%	Feedstock Supply	84%
Polymer Production	19%	Feedstock Supply	11.9%	Polymer Production	9%
Article Production	12%	Article Production	10.4%	End of Life	6.5%
End of Life	11.7%	Distribution	0.5%	Article Production	1%
Distribution	0%	End of Life	-3.4%	Distribution	0%
Particulate Matter					
Life cycle stage	Contrib.				
Feedstock Supply	63%				
Polymer Production	34%				
Article Production	5%				
Distribution	0%				
End of Life	-1.7%				
S7 – Bio-based PBS					
Climate Change		Resource Use - fossils		Land Use	
Life cycle stage	Contrib.	Life cycle stage	Contrib.	Life cycle stage	Contrib.
Polymer Production	79.6%	Polymer Production	82.1%	Polymer Production	100.4%
Article Production	20.4%	Article Production	26.8%	Article Production	1.5%
Distribution	2.0%	Distribution	1.3%	Distribution	0.2%
Feedstock Supply	0.0%	Feedstock Supply	0.0%	Feedstock Supply	0.0%
End of Life	-2.0%	End of Life	-10.2%	End of Life	-2.1%
Particulate Matter					
Life cycle stage	Contrib.				
Polymer Production	88.7%				
Article Production	16.5%				
Distribution	1.5%				
Feedstock Supply	0.0%				
End of Life	-6.7%				

## 2 6.7.3 Interpretation of characterised results

3 In this section, the characterised potential impacts of the assessed scenarios (Figures  
4 6.8-6.10 and Table B.3.8 in Annex B.3) are compared to evaluate the effects of using an  
5 alternative feedstock sources or materials for a specific fossil-based polymer of reference  
6 used in trays for food production. Therefore, the impacts of partially recycled PET and  
7 partially bio-based PET trays are compared with those of virgin PET trays. Due to their  
8 oxygen barrier properties, PLA, starch-based and Bio-PBS trays can be considered to be  
9 mainly intended as potential replacement for PET trays, which are thus considered as the



main reference for comparison. The impacts of reference scenarios (fossil-based PET and PP trays) were not compared in this case, since the two materials has quite different (oxygen) barrier properties, and they were hence assumed to be used in different applications in the market.

The comparison focuses on the sixteen, default impact categories considered in the Environmental Footprint (EF) context and in this assessment. Potential impacts on Biodiversity are discussed separately, as being estimated via an endpoint indicator that is not recommended in the PEF framework, and partially relying on different impact assessment methods for the underlying midpoint-level impact categories. In the following comparison, differences between scenario impacts lower than 10% were not considered significant, in light of the uncertainty associated with the life cycle and impact assessment models. Moreover, any comparative considerations related to the Human Toxicity and Ecotoxicity impact categories, Water Use and Land Use need to be interpreted in light of the higher uncertainty of the underlying impact assessment models and results. Finally, it has to be reminded that the results discussed below refer to a functional unit requiring the achievement of a comparable oxygen barrier performance (and thus reflecting the oxygen transmission rate of each material as relevant technical property).

The use of 24% recycled PET (alternative polymer 1) in trays manufacturing improves the environmental performance compared to virgin PET trays in all impact categories, except Ozone Depletion (which shows a six-fold increase), as well as Land Use and Resource Use – minerals and metals (where both alternatives are comparable). Leaving these exceptions apart, an impact reduction ranging from 13% (Ionising Radiation) to 43% (Ecotoxicity – freshwater) is observed, mainly due to the reduced impacts from the Polymer Production and Feedstock Supply stages. In the two most relevant categories, i.e. Climate Change and Resource Use – fossils, the impact is reduced by 26% and 34%, respectively, compared to fossil PET trays. It must be noted, however, that a higher recycled content would result in even larger impact reductions compared to the use of virgin fossil-based material (which is explored in a sensitivity analysis; Section 6.7.7.1). On the other hand, an increased Ozone Depletion impact would occur (although results for this category are less stable compared to the other established impact categories discussed above). The main reason of this substantial increase is attributed to the PET recycling process, and more specifically to the emissions of ozone depleting substances from the life cycle of sodium hydroxide and (to a lower extent) of other chemicals used in secondary PET production (with the former being responsible for more than 90% of the increased Polymer Production impacts).

Replacing fossil-based PET with partly bio-based PET in trays manufacturing, i.e. replacing fossil-based MEG with bio-based MEG from Brazilian sugarcane in PET production, results in increased impacts in more than half impact categories (9 out of 16). It is worth noting the substantially higher Particulate Matter and Land Use impact of Bio-PET (nearly 14 times compared to fossil-based PET trays), as a result of the increased contribution from Feedstock Supply (i.e. sugarcane cultivation). While the actual origin of particulate emissions could not be investigated within the applied aggregated dataset, they are likely associated with the pre-harvest burning practice, which was applied to 45% of harvested sugarcane. A sensitivity analysis was performed on this assumption, which aimed at reflecting the current situation (see Section 6.7.7.2). For the other seven categories, the observed increase is lower, but still significant, especially for Eutrophication – marine (+239%), Eutrophication – freshwater (+210%), and Human Toxicity – non-cancer (+105%) Conversely, Bio-PET trays has a lower impact than fossil-based PET ones in six categories, including two of those identified as most relevant for both alternatives, i.e. Climate Change and Resource Use – fossils. However, observed impact reductions are generally moderate (13-38%), with these extremes just being covered by Climate Change and Resource Use – fossils, respectively.

Trays made of PLA (alternative polymer 2) show a (significantly) higher impact in many (10 out of 16) impact categories compared to fossil-based PET trays, despite the

estimated mass of PLA per functional unit is nearly 60% lower than PET. The largest impact increases are observed for Eutrophication – freshwater (14 times higher than PET trays), Land Use (17 times), and Ecotoxicity – freshwater (21 time), although the increase is also relevant for Eutrophication – marine (+416%), Human Toxicity – non-cancer (+318%), Water Use (+166%), and Eutrophication Terrestrial (+144%). The increased contribution of the Feedstock Supply stage is generally responsible for this picture of results. In the six remaining categories, an impact reduction ranging from 11% to 60% is observed, with common most relevant categories (i.e. Climate Change and Resource Use fossils) showing a decrease equal to 43% and 60%, respectively

A similar situation is observed also for starch-based trays, with these showing a (substantially) increased impacts than fossil-based PET trays in the majority of the categories (12 out of 16), again mainly due to the (significantly) higher impact from the Feedstock Supply stage. Note that in this case the estimated mass of starch-based polymer required per functional unit is comparable to that estimated for PET. In the common most relevant categories (Climate Change and Resource Use – fossils), starch-based trays only result in a reduced impact for Resource Use- fossils (-60%), while their Climate Change impact is 15% higher.

As for bio-based PBS trays (which requires a nearly 40% lower mass per functional unit compared to fossil-based PET trays), the impact is also reduced by 18-96% with respect to the latter in many impact categories (11 out of 15). These also include the common most relevant categories among both alternatives, i.e. Climate Change (-61%) and Resource Use – fossils (-76%). However, Bio-PBS trays imply a higher impact in terms of Land Use (19 times), Eutrophication – freshwater (8 times), Human toxicity – non-cancer (32 times), Ecotoxicity – freshwater (4 times), and Eutrophication – marine (+51%). It has to be reminded that these results refer to a process (i.e. Bio-PBS production) that may undergo further optimisation and improvement in the future. On the other hand, the inventory applied to model such process mostly relies on literature (and partly proxy) data, which may omit relevant environmental burdens. Therefore the results relate to this alternative need to be interpreted with caution.

Focusing on estimated potential impacts on biodiversity, partially recycled PET trays show a moderately reduced impact (-24%) compared to their virgin, fossil-based counterpart. This reflects the lower burdens from Feedstock Supply and Polymer Production (which are less resource- and emission-intensive), rather than actual differences in supply chain drivers. Conversely, all trays relying on bio-based polymers (except for Bio-PBS), result in higher biodiversity impacts than fossil-based PET trays, especially in the case of starch-based ones. This is due to the increased impact from Feedstock Supply, which in the case of starch-based trays is dominated by terrestrial ecotoxicity (endpoint) impacts from starch potatoes and wheat grain cultivation. For Bio-PET and PLA trays, land occupation impacts from sugarcane or maize growing play a major role, instead. As for Bio-PBS trays, the respective biodiversity impact is nearly comparable to that of fossil-based PET trays (-14%).

As a last comment, it is noted how the Climate Change impact of Bio-PET, PLA, and starch-based trays is not significantly reduced when the contribution of non-released biogenic carbon after 100 years from landfilling is taken into account. Indeed, a decrease between 1% (starch-based trays) and 8% (PLA trays) is observed, while for Bio-PET trays the reduction equals 2%. This can be explained by the only partial biogenic carbon content in some of these polymers (Bio-PET and starch-based), and by the modest share of landfilling in the applied EU-average scenario (30%). For Bio-PBS trays the relative reduction is more relevant (-17%), reflecting the lower Climate Change impact of this alternative compared to the other examined bio-based alternatives.

#### **6.7.4 Interpretation of normalised and weighted results**

This section briefly compares the investigated scenarios based on the respective total impact scores, calculated by aggregating normalised and weighted impact assessment results across all impact categories (Table B.3.10 in Annex B.3). The comparison is

carried out according to the same criteria outlined in Section 6.7.3 in terms of contrasted trays for food scenarios, and considering significant only impact differences larger than 10%. While affected by greater uncertainty (from the calculation and application of normalisation and weighting factors), total impact scores provides a more immediate and synthetic indication of the overall environmental performance of the analysed product scenarios. This facilitates the interpretation compared to looking at characterised results from several impact categories. However, such estimates are inevitably affected by underlying values choices in the definition of weighting factors establishing an order of relevance for the different impact categories in a European decision context.

The comparison among total normalised and weighted scenario impacts reflects the overall picture of results emerged when focusing on characterised impact assessment results in single impact categories, and can be summarised in the following points:

- Partially recycled PET trays show a moderately improved environmental performance compared to virgin, fossil-based PET trays, with 30% lower total impact score. This reduction is almost proportional to the assumed recycled content (i.e. 24%).
- Partially bio-based PET trays (relying on MEG derived from Brazilian sugarcane) shows a slightly worse performance compared to their fossil-based counterpart, with a total impact score that is 23% higher than the latter.
- Biodegradable PLA and Bio-PBS trays show a reduced overall impact compared to fossil-based PET trays, especially in the case of Bio-PBS (60% lower impact score, in light of a 40% lower polymer demand per functional unit). For PLA the reduction is modest (26%) despite the nearly 60% lower polymer demand per functional unit). The total impact of biodegradable starch-based trays is instead comparable with that of fossil-based PET trays (6% reduction).
- Compared to partially recycled PET trays, biodegradable Bio-PBS trays still show a reduced impact score (-44%), while starch-based trays show a moderately increased overall impact (34%). PLA trays are instead comparable to partially recycled PET trays (5% increase in the total impact).

### 6.7.5 Effects of indirect land use change (iLUC)

When the contribution of GHG emissions from iLUC is taken into account, only a small changes are identified in the Climate Change impact of bio-based alternatives (i.e. Bio-PET, PLA, starch-based, and Bio-PBS trays) (Figure 6.8 and Table 6.13). A maximum increase by 6% is observed for bio-based PBS trays (which have the lowest Climate Change impact when iLUC is not considered), while for other bio-based alternatives the increase ranges between 0.5% (starch-based trays) and 3% (PLA trays). This reflects the only partial bio-based content in the polymer (Bio-PET and starch-based trays), or the relatively higher Climate Change impact even when iLUC is not considered (PLA trays).

Overall, accounting for iLUC effects does not affect the relative performance of the assessed scenarios (i.e. their comparison is not affected). However, this outcome needs to be read also in light of the GHG emission factors applied for iLUC in this assessment, which appear to fall in the lower end of the range of values available in the recent literature (see Section 4.4.16.3 in the Method (Report I) for additional comments).

### 6.7.6 Alternative End of Life options

This section discusses characterised scenario results calculated by individually applying each viable End of Life option as an exclusive (100%) End of Life scenario (as presented in Figures 6.12 to 6.14). The main purpose is to evaluate how the performances of the single trays for food scenarios are affected by the change in the applied End of Life option, rather than to mutually compare such scenarios for different End of Life options. However, the results should not be interpreted as a strict comparison among different End of Life alternatives for the studied article, since the evaluation applies a product

perspective, and burdens/benefits of selected End of Life options (e.g. recycling) are partitioned between different product life cycles (which prevents from capturing the full implications of having a given waste stream routed to a specific End of Life option). In a waste management system perspective, where allocation is not needed, the total (system-wise) savings associated with the End of Life pathway “100% recycling” would likely be higher than what presented in this report where a product perspective is applied.

Because of the generally modest contribution of the End of Life stage, in several impact categories no relevant changes in the overall scenario impacts are observed with the change in the applied End of Life option. This is especially the case of bio-based scenarios (which have relatively higher impacts), while for scenarios relying on fossil-based polymers (both virgin and recycled) and on Bio-PET, larger differences can be observed, at least for some impact categories and End of Life options. Keeping in mind this possibly limited range of variation, the following considerations can be made.

None of the five considered product End of Life options (i.e. recycling, incineration, landfilling, and for biodegradable polymers, composting and anaerobic digestion) can be identified as preferable across all the assessed impact categories and trays for food scenarios. For fossil-based PET trays (both virgin and partially recycled), 100% mechanical recycling is the best End of Life scenario in most impact categories (i.e. all except Eutrophication – freshwater, Ionising Radiation, Ozone Depletion and Resource Use – minerals and metals). For partially bio-based PET trays, a similar situation is observed, although in this case six impact categories are excluded rather than only four (i.e. Ionising Radiation, Ozone Depletion, Climate Change, Photochemical Ozone Formation, Resource Use – fossils, and Particulate Matter). This is likely due to the assumed substitution of mostly fossil-based PET as a virgin material (which in most of the mentioned categories has lower production impacts and hence benefits than bio-based PET). For PP trays, the mechanical recycling scenario is the most preferable in an even more limited number of categories (i.e. seven), while for trays made of biodegradable polymers recycling was not assessed, as not being a viable End of Life option as of today. Only in Ozone Depletion recycling is found to be the worst scenario for all the assessed fossil-based tray alternatives, while for Bio-PET trays this is also the case of Climate Change, Ionising Radiation, Photochemical Ozone Formation, Resource Use Fossils and Particulate matter. In the (few) remaining categories, recycling has an intermediate performance between incineration and landfilling.

For fossil-based PET trays (both virgin and partially recycled) and Bio-PET ones, the 100% incineration scenario is only preferable in those four/six categories where recycling is not preferred (as listed above). This also applies to PP trays, although in this case the number of categories where incineration is preferable is higher (i.e. nine). For all these alternatives, the incineration scenario shows the worst performance only in terms of Climate Change (except for Bio-PET trays, where recycling is the worst scenario) and Water Use. As for trays made of biodegradable polymers, a diverse situation is observed, depending on the polymer. For starch-based trays, the incineration scenario is the most preferable in the majority of impact categories (12 out of 16), while for PLA and Bio-PBS trays this occurs in nearly half of the categories (i.e. seven and six, respectively). Conversely, incineration shows the worst performance only in a limited number of categories, i.e. Climate Change (PLA and Bio-PBS trays), Human Toxicity – non-cancer (PLA and starch-based trays), and Water Use (Bio-PBS trays).

In many impact categories, 100% landfilling is the worst-performing scenario for several food tray alternatives, in line with the priority order outlined in the “Waste Hierarchy”, which sets disposal as the least preferable option (EC, 2008). However, for some biodegradable alternatives (which miss the recycling option), the landfilling scenario is preferable (or comparable to the best-performing End of Life scenario) in few impact categories, including Climate Change (Bio-PBS trays), Human Toxicity – cancer (PLA and starch-based trays), Ecotoxicity – freshwater (starch-based trays), and Water Use (PLA trays). Moreover, if the contribution of the missed release of biogenic carbon after 100

years from landfilling of bio-based polymers is taken into account, 100% landfilling is the preferable scenario for bio-based PET and starch-based trays, while for PLA and Bio-PBS trays other scenarios are still preferable.

For biodegradable trays (i.e. PLA, starch-based and Bio-PBS trays), composting and anaerobic digestion were also assessed as exclusive End of Life scenarios (except Bio-PBS trays, for which only composting was considered). In several impact categories (10 out of 16), 100% composting is indeed the preferable scenario for Bio-PBS trays, while for PLA and starch-based trays this is the case of only a limited number of categories (i.e. seven at the most, if the few categories where composting is comparable to the best-performing anaerobic digestion scenario are also included). Conversely, for these two alternatives, composting is the worst scenario (or is comparable to the worst-performing anaerobic digestion scenario) in many impact categories, (i.e. nine for PLA trays and twelve for starch-based ones). For Bio-PBS trays, this occurs in five categories. 100% anaerobic digestion of PLA trays is preferable only in four categories, although there are other three categories where it is essentially comparable to the best-performing composting scenario. For starch-based trays, this is only the case of a very limited number of categories (i.e. three). In the remaining categories, for both PLA and starch-based trays anaerobic digestion shows the worst performance, or is comparable with the worst-performing composting scenario.

#### **6.7.7 Interpretation of results related to macro- and micro-plastics generation**

This section discusses the results presented in Section 6.6.3 on the estimated potential generation of macro- and micro-plastics of the assessed trays for food scenarios. The latter are compared according to the same criteria applied to the interpretation of the potential environmental impacts (Section 6.7.3), initially focusing on the results from the two-level approach applied as a base case. The results from the approaches applied as a sensitivity analysis are discussed separately.

Regarding the total loss of plastics (macro- and micro-plastics) and the resulting release to ocean and to the terrestrial environment (Figure 6.16), the two base-case approaches (PLP method and Expanded PLP method) provide similar results when comparing the different trays for food scenarios. In both cases, the scenario considering the use of PLA as alternative feedstock sources for PP trays show the best performance (-45%) in the total loss and release compared to the respective fossil-based counterpart. Alternative scenarios to PET trays show no significant difference (<1%). While bio-PBS as alternative to PP trays also shows reduced plastic loss and release (-15%), starch-based alternative scenario results in a larger plastic loss and release (+55%). This is mostly a consequence of the amount of polymer required per functional unit for the different scenarios, which leads to a proportionally reduced or increased loss and release of plastics from both upstream life cycle stages (micro-plastics) and especially from product End of Life (macro-plastics). Note, however, that the applied methods rely on mass-based indicators (which are affected by differences in tray mass) rather than quantifying the number of items (or plastic fragments/particles) lost or released to the environment. If this was the case, all scenarios would result in the same macro-plastics generation at End of Life when applying the PLP method, since trays with a lower mass would not imply a reduction in the contribution of product littering and waste mismanagement (which are based on release rates that are not affected by the type of feedstock or material).

Comparing the two base-case approaches, the Expanded PLP method results in a higher plastic loss and release to ocean and the terrestrial environment compared to the original PLP method, equalling between 36% and 49% for fossil-based scenarios (Figure 6.16). This is mainly due to an increased contribution of macro-plastics loss at End of Life, which also accounts for mismanagement of plastic waste exported for (sub-standard) recycling outside the EU (where the overall waste mismanagement share is increased by 2 - 6% compared to the assumption that all plastic waste is recycled in Europe). In the case of bio-based scenarios, this increase is not observed as trays are composted instead

of recycled at the End of Life. Conversely, the two additional micro-plastics sources considered in this alternative approach (road markings and marine coatings) does not significantly affect the results, due to the overall modest contribution of micro-plastics to the total release (in terms of mass).

Focusing on the relation between total loss and release (which depends on the parameters of the PLP method), the overall plastic loss is considered a release to the environment. In relative terms, release to ocean accounts for a lower share of the plastic loss (25%), while release to the terrestrial environment is larger (75%). As shown in Figure 6.18 for the PLP method, the total plastic release to both ocean and the terrestrial environment is dominated by the release of macro-plastics at the End of Life (more than 99%), while micro-plastics released throughout the upstream life cycle have only a minor role. Similar results are also observed for the Expanded PLP method (not shown), despite two additional sources of micro-plastics are considered in the latter (i.e. road markings and marine coatings). In both cases, this is a consequence of the prevailing mass of macro-plastics, which is directly related to the mass of trays required per functional unit (reference flow), by means of higher loss and release rates compared to micro-plastics. Therefore, the mass of macro-plastics is at least one order of magnitude higher than the mass of micro-plastics, which instead depends on the quantity of relevant lifecycle process by means of (much) lower loss and release coefficients (depending on the source).

Due to the prevailing role of macro-plastics, the comparison among the sole macro-plastics release to ocean (from product littering and waste mismanagement) estimated for trays for food scenarios with both base-case approaches reflects the results obtained when looking at the total plastic release into such compartment (Figure 6.17). Comparative results are also in line with those of the sensitivity analysis based on marine litter rates, where PLA trays still show the lowest release. However, the sensitivity analysis estimate smaller amount of plastic release to ocean due to the limited granularity of the database sourcing consumption intensity data. However, it must be reminded that all the approaches rely on mass-based indicators, and the resulting trends are thus determined by the mass of material required to fulfil the functional unit. This implies that scenarios relying on lighter trays provide a lower contribution to macro-plastics generation (and vice versa), which would not be the case if an indicator based on the number of released items was considered. Note, however, that such estimates only provides information at the level of product category (i.e. trays) without accounting for specific product properties (e.g. type of polymer) and for the specific supply-chain configuration (e.g. in terms of type and location of waste treatment operations). For all scenarios, emissions from product littering represent around a third of the total macro-plastics release, between 12% and 18%, highlighting the important role of waste mismanagement.

With regard to the origin of released plastics (in terms of life cycle stage and source), macro-plastics entirely originate from the End of Life stage (Figure 6.19), as they are only derived from product littering or mismanagement of the product as waste. On micro-plastics release, the relative contribution of the different sources (e.g. loss of plastic pellets and tire abrasion) depends on the characteristics of the supply-chain (e.g. feedstock origin, which affect the contribution from transport across the life cycle), as well as on the mass of polymer required per functional unit (which affect, to different extents, all the considered sources). However, within the limited role of micro-plastics in the overall release to the environment, the most relevant contribution is generally provided by pellet losses from the Polymer and Article Production stages, while transport-related sources have a more restricted role.

In conclusion, due to the dominant role of macro-plastics loss and release at the End of Life of trays for food, a best overall performance is associated to the scenario requiring a lower amount of plastic to fulfil the functional unit (i.e. relying on lighter trays; mass-driven results). In this context, the use of PLA trays is the scenario with the lowest plastic loss and release to both ocean and the terrestrial environment. The same

considerations apply to the sole release of macro-plastics to ocean at End of Life, although in this case the use of an indicator based on the number of items (i.e. trays) released would provide the same result regardless of the scenario when applying the PLP method or the sensitivity estimates based on marine litter rates. Trays with different masses would indeed equally contribute to littering and waste mismanagement at End of Life. Conversely, an items-based indicator would still show different results for fossil-based trays in the Expanded PLP method, with a higher release of macro-plastics due to mismanagement of waste exported for recycling in non-EU countries. From a methodological perspective, the different approaches provide similar results when comparing alternative scenarios.

### **6.7.8 Sensitivity analysis**

A sensitivity analysis has been performed on a number of relevant parameters or assumptions, to evaluate the effects of their variation on the potential impacts of the affected scenario(s), and on the comparative LCA results. The following aspects have been specifically considered:

1. Recycled content in R-PET trays;
2. “Slash and burn” rate for sugarcane used as a feedstock for Bio-PET trays.

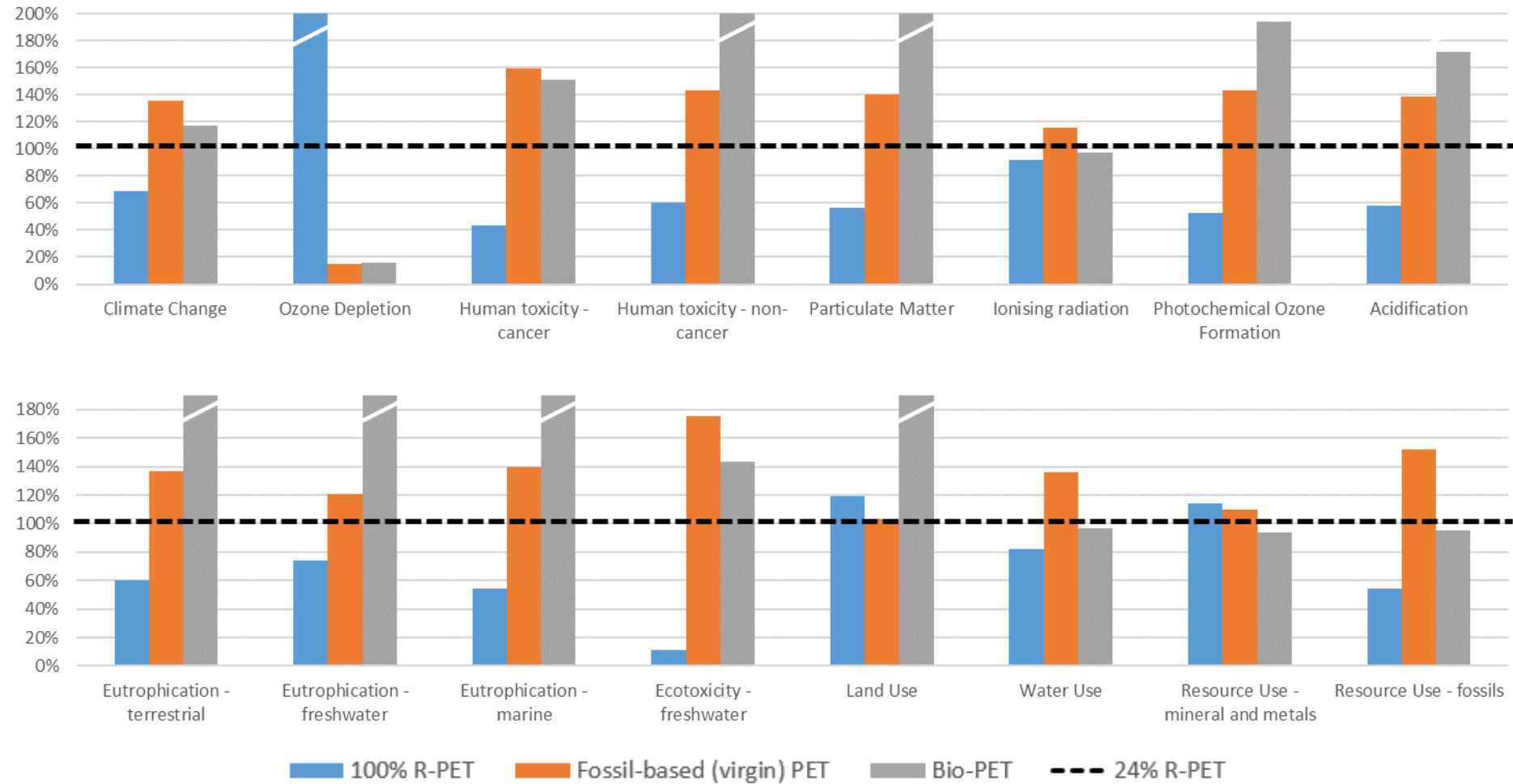
The following sections present the outcome of the sensitivity analysis for each of the aspects above. The results are reported by comparing recalculated impacts of the affected scenario(s) with those of the respective base case situation, of the corresponding reference scenario, and of the alternative scenarios evaluated for the same reference material.

#### **6.7.8.1 Recycled content in R-PET trays**

This section explores the effects of increasing the recycled content of PET trays for food to 100%. As a base case, this parameter was estimated to reflect the current average recycled content at the EU level, equalling 24%. However, from a technical point of view, a higher recycled content could be achieved, potentially reaching 100% for PET trays (in analogy with PET beverage bottles, although no real examples are available on the market in this respect). A sensitivity analysis was thus performed to assess the effects of this possibility.

The results of the analysis are presented in Figure 6.20. Scenario impacts are expressed as a function of the impacts of the base case of the sensitivity scenario (i.e. 24% R-PET trays), which is set as a 100% baseline. This allows to show multiple impact categories expressed with different units on the same chart.

An increased content of secondary material significantly reduces the impact of recycled PET trays in all except three of the assessed categories, with a decrease ranging from 8% (Ionising Radiation) to 88% (Ecotoxicity - freshwater) compared to the base case. In all these categories, recycled PET trays now outperforms to a larger extent virgin PET trays, with an impact reduction ranging from 21% to 93% compared to the latter. However, with a 100% recycled content, an impact increase is observed in those categories where recycled PET trays already showed a worsened or comparable performance with respect to virgin PET trays, i.e. Ozone Depletion (+200% compared to 24% R-PET trays), Land Use (+20%) and Resource Use – minerals and metals (+14%). In these categories, increasing the content of secondary material thus makes the use of recycled PET even less beneficial compared to its virgin, fossil-based counterpart. Compared to partly bio-based PET, a worsened performance is only observed for Ozone Depletion and Resource Use – minerals and metals, while for the remaining impact categories, 100% recycled PET trays are always preferable (15-97% impact reduction), except for Ionising Radiation, where both alternatives are comparable.



**Figure 6.20.** Results of the sensitivity analysis on the recycled material content in PET trays for food, increased from 24% to 100%. Scenario impacts are expressed as a percentage of the base case of the sensitivity scenario (i.e. 24% R-PET trays), which is set as a baseline. Note that in some impact categories a part of the results is out of scale and is curtailed.



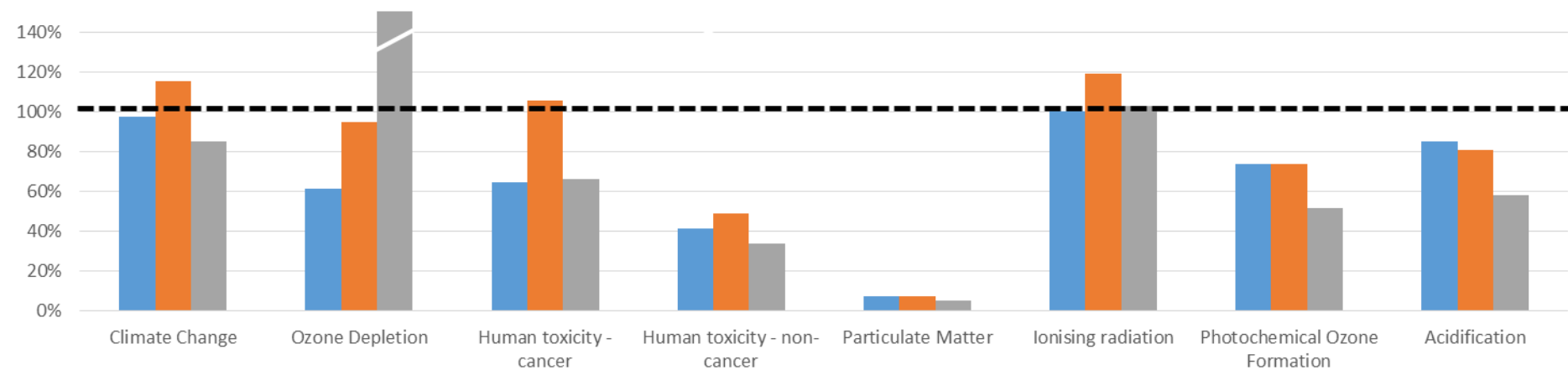
#### **6.7.8.2 *Slash and burn” rate for sugarcane***

This sensitivity analysis explores the effects of completely removing the “slash and burn” practice in the cultivation of Brazilian sugarcane used as a feedstock for bio-based PET production. 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 “slush and burn” rate was reduced from 45% (assumed as a base case) to 0%.

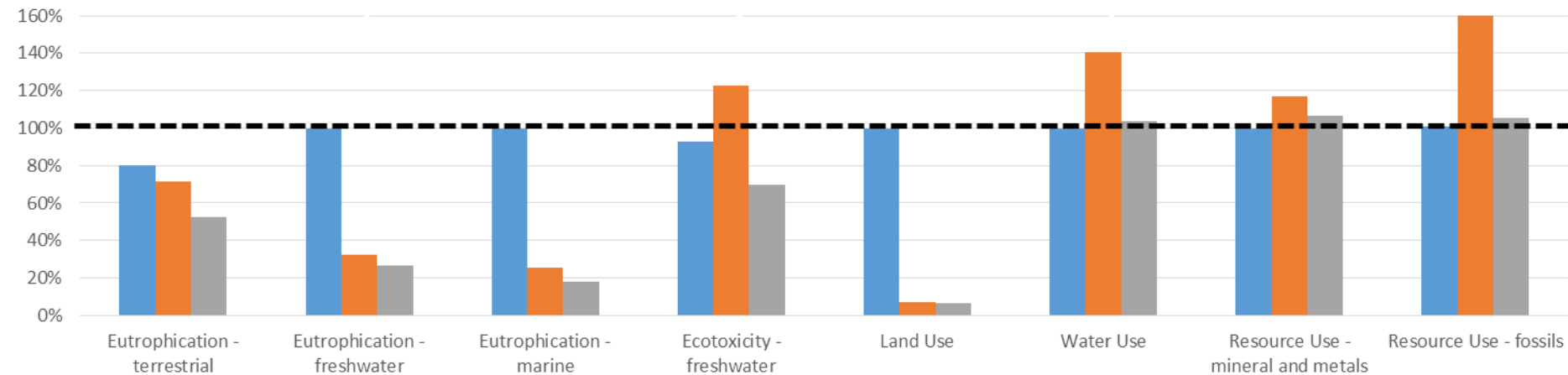
Figure 6.21 shows the results of the sensitivity analysis for bio-based PET trays. Scenario impacts are expressed as a function of those of the base case of the sensitivity scenario (i.e. bio-based-PET trays relying on Brazilian sugarcane cultivated with a “slash and burn” rate of 45%), which is set as a 100% baseline.

Completely phasing out the sugarcane pre-harvest burning practice decreases the impact of bio-based PET trays in seven impact categories (15-92%), while no or irrelevant changes are observed in the remaining ones. The largest reductions are observed in Particulate Matter (-92%), Human Toxicity – non-cancer (-59%), Ozone Depletion (-39%), Human Toxicity – cancer (-36%) and Photochemical Ozone Formation (-26%). In light of these reductions, the comparison with the reference scenario (i.e. fossil-based PET trays) is moderately improved, with the latter now being outperformed in more than half (nine) impact categories instead of six, although the achieved impact reduction are modes (15-39%). Moreover, Bio-PET trays still show a substantially larger impact in terms of Land Use, Eutrophication – marine, and Eutrophication – freshwater.

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Bio-PET (0% slash & burn)   Fossil-based PET   24% R-PET   - - - Bio-PET (45% slash & burn)

4

**Figure 6.21.** Results of the sensitivity analysis on the sugarcane “slash and burn” rate for bio-based PET trays for food. Scenario impacts are expressed as a percentage of the base case of the sensitivity scenario (i.e. bio-based PET trays from Brazilian sugarcane with 45% “slash and burn” rate), which is set as a baseline. Note that in some impact categories a part of the results is out of scale and is curtailed.

6

## **7 Case study 4: Agricultural mulching film**

This case study assesses plastic mulching film used in agriculture. Plastic mulching is an agricultural practice consisting in the mechanical application of a plastic film (black, white or transparent, with low thickness) on the soil. Seedlings and young plants are planted by punching holes in the film. The primary function of plastic mulching films is seedlings and shoots protection, by insulating and preventing evaporation (Tarara, 2000), but it is also used to reduce exposure to pests (McKenzie and Duncan, 2001). Additional advantages of plastic mulching are recognized, such as yield increase, earlier development of seeds and fruits, weeds control, and a consequent reduction in fertilisers and pesticides input (Scarascia-Mungozza, 2011; Chalker-Scott, 2007). Thanks to the economic and agronomic benefits it brings to cultivation, it has become a widely used technique in the global agriculture. However, the use of plastic mulches may also have adverse effects on soil quality and the environment, especially where proper and complete collection after use is not ensured. This promotes the accumulation of plastic fragments in soil, which in the long term may negatively affect soil-quality and crop growth. Moreover, plastic residues may further disintegrate into micro-plastics, which can act as a media where agro-chemicals are adsorbed (Steinmetz et al., 2016) with potential detrimental effects on soil and/or other ecosystems (although these effects are not yet proven nor understood). Plastic additives may also be released (both during and after use), and pesticide runoff may be promoted (while potentially reducing their leaching). Finally, plastic mulches may increase degradation and depletion of soil organic matter during use (due to higher temperatures), promoting the subsequent release of greenhouse gases and reduction in the water retention capacity of soil (Steinmetz et al., 2016).

In Europe, plastic mulching is used mainly in horticulture. Due to its high flexibility and durability, 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-Mungozza, 2011).

The focus of this study is on generic mulching film used in large-scale horticulture, with no specification of colour or crop-specific functions. This choice allows to keep a broader scope, extend the validity of results beyond very specific situations, and overcome the scarcity of information on the characteristics of mulching film applied to specific (horticultural) crops.

### **7.1 Assessed scenarios**

The potential life cycle impacts of mulching film manufactured from different polymers and/or feedstock sources were explored by modelling a number of scenarios considering the use of common alternatives currently available on the market. The different types of polymer and feedstock considered in the scenarios selected for this case study are summarised in Table 7.1, which also specifies the End of Life options analysed for each scenario. Conventional non-biodegradable mulching film produced from fossil-based (LDPE) was defined as the reference scenario (S1). The use of recycled LDPE from post-consumer plastic waste as a feedstock was also assessed, by assuming 100% of the material input to film manufacturing coming from recycling (S2). Even though recycled material can be mixed with virgin material at different shares, the analysis focused on mulching film entirely relying on recycled polymer input, with the aim of assessing the effects of a complete substitution of the virgin material (and in the absence of representative data on the current average recycled content at the EU level). It is acknowledged that this is an optimistic scenario for the current situation.

Two partially bio-based and biodegradable alternatives were explored in scenario 3 (S3) and in scenario 4 (S4), where mulching film is manufactured from polymer blends combining a biomass-derived polymer with a fossil-based co-polymer. In particular, in S3 a starch-based polymer is considered, which is obtained by blending Thermoplastic

1 Starch (TPS; 40% of the blend) and PBAT<sup>114</sup> (for the remaining 60%). This is referred to  
2 as Starch-based polymer in the following. In S4, a PLA-based polymer is considered,  
3 where Polylactic Acid (PLA) is still mixed with PBAT<sup>1</sup> (according to a 45%-55%  
4 proportion).

5 For PLA production, US maize was considered as a feedstock, since nearly 90% of PLA  
6 currently consumed in Europe comes from the US (Eurostat, 2019a), where maize is  
7 used as a source of starch-derived sugar (dextrose) by the largest producer in the  
8 Country (Vink and Davies, 2015). Conversely, starch used in starch blend production was  
9 derived from the current EU-average mix of starch crops, being one of the major  
10 producers of starch blends located in Europe. The mix includes maize (47% on starch  
11 basis), wheat (40%) and potatoes (13%) (Starch Europe, 2019). PBAT used in both  
12 types of blend is based on fossil sources and no agricultural feedstock is needed in this  
13 case.

14 Regarding End of Life, all currently viable treatment and disposal options were considered  
15 for each mulching film material. For films made of conventional, non-biodegradable  
16 polymers (i.e. LDPE and recycled LDPE) viable options that were considered include  
17 mechanical recycling, incineration and landfilling. For biodegradable alternatives (i.e.  
18 starch-based and PLA-based film), only in-situ (i.e. in- and on-soil) biodegradation was  
19 considered, since they are primarily intended to be left on the field after the growing  
20 season to biodegrade on soil (or into the soil after being ploughed back before the next  
21 season). As a base case, the impacts of each scenario were assessed with reference to  
22 an EU-average End of Life scenario including all the options currently applied at the EU  
23 level for the specific material. These include all the viable options reported above, which  
24 for non-biodegradable films were combined as described in Section 7.4.5.1 on End of Life  
25 modelling. In addition, scenario impacts were calculated by individually considering each  
26 viable End of Life option.

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<sup>114</sup> PBAT: Polybutylene Adipate co-Terephthalate.

**Table 7.1.** LCA scenarios assessed for the mulching film case study.

Scenario	Polymer	Monomer or Co-polymer	Feedstock	End of Life options <sup>(1)</sup>
1 - Conventional polymer (reference)	LDPE <sup>(2)</sup>	Ethylene	Fossil-based (crude oil/natural gas)	Recycling Incineration Landfilling
2 - Alternative polymer 1	R-LDPE (100% recycled content)	Ethylene	Waste LDPE (post-consumer)	Recycling Incineration Landfilling
3 - Alternative polymer 2	TPS/PBAT blend <sup>(3,4)</sup>	TPS 1,4-Butanediol Adipic acid PTA <sup>(5)</sup>	EU mix of starch crops <sup>(6)</sup> Crude oil/natural gas	In-situ biodegradation
4 - Alternative polymer 3	PLA/PBAT blend <sup>(7,8)</sup>	Lactic acid 1,4-Butanediol Adipic acid PTA <sup>(5)</sup>	Maize (US) Crude oil/natural gas	In-situ biodegradation

<sup>(1)</sup> The impacts of scenarios were individually assessed for each listed End of Life option, as well as for a combination of such options reflecting as far as possible the current average situation at the EU level.

<sup>(2)</sup> LDPE: Low Density Poly Ethylene.

<sup>(3)</sup> TPS: Thermoplastic Starch; PBAT: Polybutylene Adipate co-Terephthalate.

<sup>(4)</sup> A 40% Starch and 60% PBAT blend is considered, according to the information received from stakeholder consultation, reporting that renewability of starch-based polymers can reach 40% (although when used in packaging applications).

<sup>(5)</sup> PTA: Purified Terephthalic Acid.

<sup>(6)</sup> Maize (47%), Wheat (40%), Potatoes (13%), in terms of starch product equivalents (Starch Europe, 2019).

<sup>(7)</sup> PLA: PolyLactic Acid; PBAT: Polybutylene Adipate co-Terephthalate.

<sup>(8)</sup> 45% PLA and 55% PBAT, according to the technical specification of a major biodegradable PLA blend available on the market.

## 7.2 Functional Unit and reference flow

The main function of plastic mulching film is to protect crops and facilitate their growth by controlling weeds, by retaining the moisture content in soil and roots, and by maintaining soil temperature (Razza and Cerutti, 2017). The functional unit was thus defined as: "providing mulching to one hectare of agricultural land in Europe, cultivated with horticulture crops (e.g. melon, strawberry, zucchini etc.), for a period corresponding to an average growing season (i.e. four to five months)" (Table 7.2).

1 **Table 7.2.** Definition of the functional unit for mulching film LCA scenarios.

Aspect	Description
<b>“What”</b> (function(s) or service(s) provided)	Providing mulching for agricultural land cultivated with horticulture crops through plastic film
<b>“How much”</b> (extent of the function(s) or service(s))	One hectare of cultivated land (i.e. 0.6 ha of mulched land)
<b>“How well”</b> (expected level of quality)	Adequately protecting the crop and facilitating its growth by controlling weeds, by retaining the moisture content in soil and roots, and by maintaining soil temperature
<b>“How long”</b> (duration/lifetime of the function or service)	For one entire growing season (i.e. four to five months on average)
<b>“Where”</b> (location/geography of the service)	In Europe

2  
3 The reference flow (i.e. the amount of polymer required to fulfil the functional unit) was  
4 calculated based on the thickness of the film in the specific scenario, the net area of land  
5 to be mulched, and the density of the relevant film material (Table 7.3). At this regard, it  
6 is noted that it was not possible to identify a thickness ensuring that mulching films made  
7 from different polymers provide comparable agronomic performances. Therefore, the  
8 typical (average) thickness of films currently available on the market for horticulture  
9 applications was considered, which varies according to the material<sup>115</sup>. As for the  
10 extension, it was assumed that mulching 1 ha of cultivated land requires on average 0.6  
11 ha of film, considering the typical space (i.e. interline) between two contiguous seedlings  
12 lines that is left un-mulched in horticulture. Finally, typical average material densities  
13 were considered, as reported in the literature or in technical specifications of  
14 commercially available mulching films.  
15

<sup>115</sup> Note that the thickness assumed for non-biodegradable LDPE mulching films (i.e. 35 µm) fulfil the requirement from EN 13655 of having a thickness higher than at least 20 µm for thermoplastic mulching film intended to be removed from soil after use for recovery (which is the case for non-biodegradable films considered in this case study).

**Table 7.3.** Calculation of the reference flow for mulching film LCA scenarios.

Material	Thickness ( $\mu\text{m}$ ) <sup>(1)</sup>	Density ( $\text{kg}/\text{m}^3$ ) <sup>(2)</sup>	Reference flow <sup>(3)</sup> ( $\text{kg}/\text{FU}$ )
LDPE; R-LDPE	35 (15-50) <sup>(4)</sup>	0.925 (0.92-0.93) <sup>(5)</sup>	194.25
Starch based polymer	12 (12-18) <sup>(6)</sup>	1.27 (1.23-1.29) <sup>(6)</sup>	91.44
PLA based polymer	10 <sup>(7)</sup>	1.38 (1.34-1.40) <sup>(7)</sup>	82.80

<sup>(1)</sup> Values in parenthesis are the extremes of the typical range of thickness in which mulching film of the specific material is available on the market.

<sup>(2)</sup> Values in parenthesis refer to the typical range of variation of the density of the specific mulching film material.

<sup>(3)</sup> 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.

<sup>(4)</sup> Source: OWS (2017).

<sup>(5)</sup> Source: Scarascia-Mungozza et al. (2011).

<sup>(6)</sup> Source: technical properties of a major starch-based mulching film available on the market.

<sup>(7)</sup> Source: technical properties of a major PLA-based mulching film available on the market.

### 7.3 System boundary

In all the assessed scenarios, the system boundary was set according to a cradle-to-grave perspective to cover the most relevant stages and processes of the mulching film life cycle, as described below and shown in Figures 7.1 to 7.4.

- *Feedstock Supply*<sup>116</sup> – covering extraction, transport and possible refining of crude oil and natural gas (fossil-based polymers), collection, transport and sorting of plastic waste (recycled polymers), crop cultivation (bio-based polymers), as well as transport of these feedstock sources to downstream conversion processes (e.g. naphtha cracking, polymer recycling, wet milling of starch crops, etc.);
- *Polymer Production*<sup>117</sup> – covering all the activities associated with the conversion of feedstock materials into the relevant monomer(s) and final polymer, including any transport among these activities and final transport of polymer granulate to downstream manufacturing processes;
- *Article Production*<sup>118</sup> – including mulching film manufacturing through blown film extrusion of polymer granulates;
- *Distribution* – including transport of mulching film from the manufacturing site to distribution centres and from these to final users;
- *End of Life* – covering removal of non-biodegradable mulching film after use and its transport to recycling, incineration or disposal (including these treatment processes and any avoided processes from virgin material or energy substitution). For biodegradable mulching film, in-situ biodegradation was included.

A different nomenclature was applied for some life cycle stages compared to the default nomenclature specified in the Method (and in the PEF framework) to make it more relevant for the investigated supply chains and the project scope. Moreover, the default “Raw Material Acquisition and Pre-processing” stage was further split into two separate sub-stages (i.e. Feedstock Supply and Polymer Production), to allow disaggregating the impacts of feedstock supply from downstream conversion processes, and hence to better

<sup>116</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

<sup>117</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF method.

<sup>118</sup> Corresponding to the default stage of “Manufacturing” specified in the Method and in the PEF framework.

1 appreciate any differences among the use of different feedstock sources (in line with the  
2 project goal).

3 It has to be noted that, for non-biodegradable mulching films, the contribution of  
4 contamination by soil particles after removal from land was considered in the transport of  
5 used mulching film to the respective End of Life treatments. Conversely, the burdens  
6 associated to the presence of soil within such treatments were only partially addressed  
7 (i.e. limited to the case of recycling and by means of modelling approximations). This is  
8 due to the lack of representative data on the average soil composition at the EU level  
9 (relevant for incineration, landfilling and treatment of wastewater from recycling), as well  
10 as to the absence of information on how the recycling process is actually affected to  
11 handle soil contamination<sup>119</sup>.

12 The Use stage (i.e. mulching film application) was also excluded from the system  
13 boundary. The burdens of this stage can indeed be considered identical in all the  
14 investigated scenarios<sup>120</sup>, and can thus be omitted from a comparative assessment, like  
15 the one performed in this study.

16 Finally, additives used in mulching film production were not included in the assessment  
17 (both in terms of production and of their possible fate and effects at End of Life) due to  
18 the lack of complete and consistent data and information in this respect. This is  
19 acknowledged as a limitation of this study, also considering that mulching film properties  
20 are normally optimised by means of additives (e.g. plasticizers, ultraviolet stabilizers,  
21 etc.) in the manufacturing stage. However, additives used for Thermoplastic Starch  
22 production (mainly plasticisers) were considered, as they represent a non-negligible and  
23 essential portion of the final polymer (i.e. 25%) and are required to ensure suitable  
24 material properties and processability. Moreover, production of additives used in starch-  
25 based polymers can account for an important share of the respective cradle-to-gate  
26 Climate Change impact and energy demand, reaching 46% for polymer grades including  
27 a share of additives in the range of 30% (Broeren et al., 2017).

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<sup>119</sup> For instance, it is not known if specific process steps are undertaken at recycling facilities for the removal of soil or any other contamination, or if this role is left to the conventional washing/flotation processes carried out during common recycling processes (which is the assumption performed in this case study, as better detailed in Section 7.4.5.3).

<sup>120</sup> The burdens from agricultural operations for mulching (film) applications normally depend on the extension to be covered rather than the mass of material to be applied.



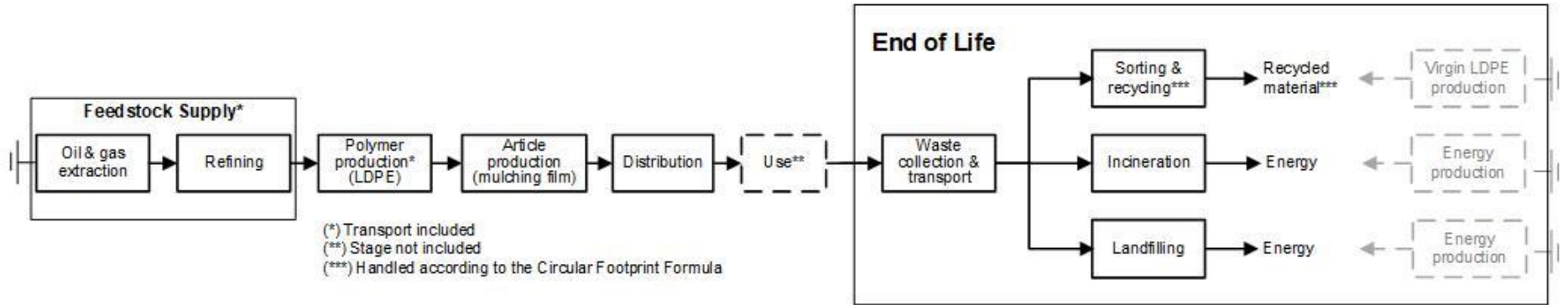


Figure 7.1. System boundary for fossil-based LDPE mulching film (Scenario 1).

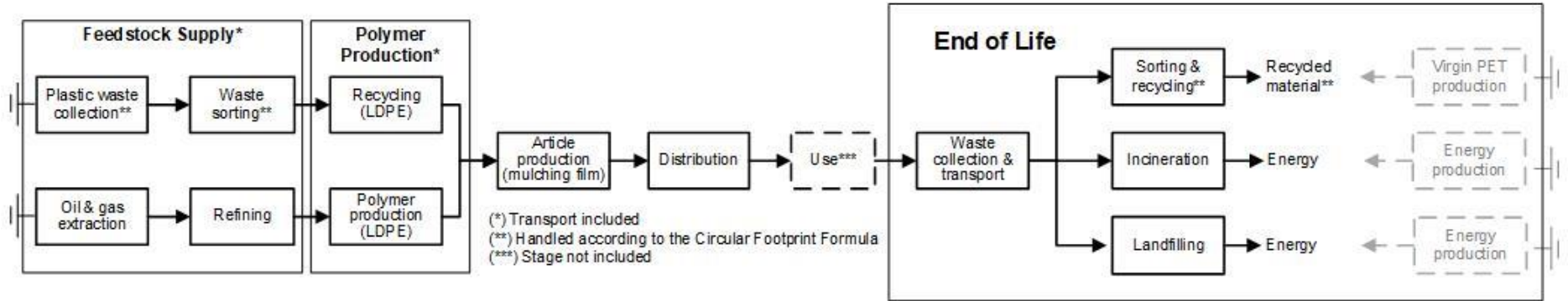


Figure 7.2. System boundary for recycled LDPE mulching film (Scenario 2).

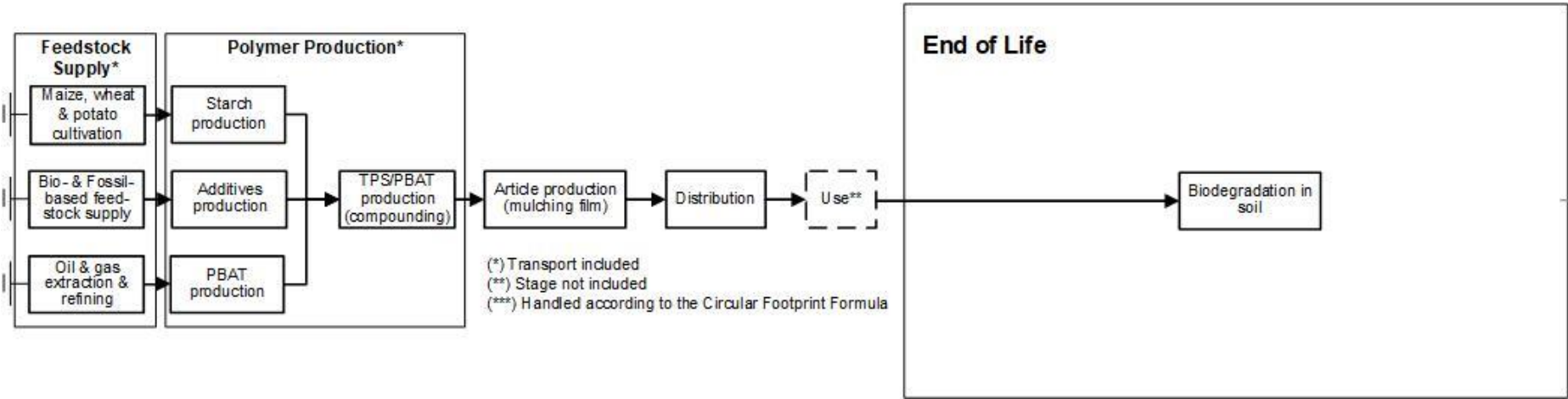


Figure 7.3. System boundary for starch-based mulching film (Scenario 3).

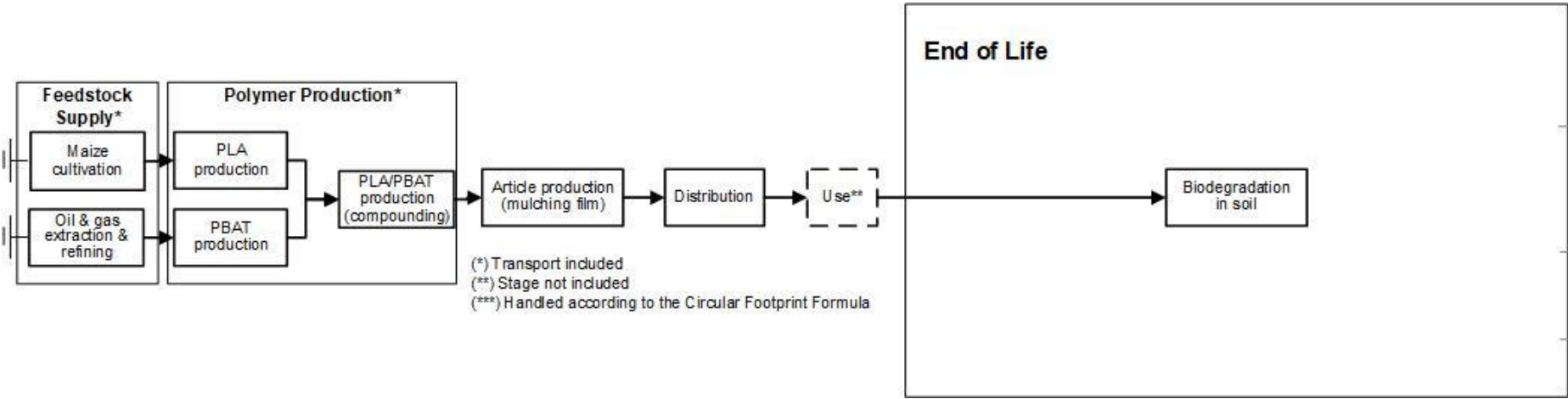


Figure 7.4. System boundary for PLA-based mulching film (Scenario 4).

## **7.4 Life Cycle Inventory**

This section describes the overall approach for developing the Life Cycle Inventory of the analysed scenarios, along with the related assumptions and data sources. The description is separated by major lifecycle stages, in the following sub-sections (7.4.1 – 7.4.6). The list of processes, related data sources, and main modelling details are provided in Tables B.4.1 to B.4.4 in Annex B.4.

### **7.4.1 Feedstock Supply Stage**

#### **7.4.1.1 Fossil-based LDPE**

For virgin, fossil-based polymers (i.e. LDPE), the stage of Feedstock Supply includes the processes of crude-oil and natural gas extraction and transport, naphtha production in crude oil refineries, and its subsequent transport to downstream conversion processes (i.e. naphtha cracking, or also catalytic reforming). The modelling of these activities was carried out as described in Section 4.4.1.1, where the reader is referred to for further details on data sources, covered sub-processes and related environmental burdens, main modelling choices (e.g. allocation) and any adjustments performed in the implementation of the applied datasets.

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, in the absence of disaggregated datasets related to 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 is hence entirely discussed in Section 7.4.2.3, focusing on Polymer Production.

#### **7.4.1.2 Recycled LDPE**

For recycled LDPE, Feedstock Supply consists of collection of post-consumer LDPE waste, and its subsequent transport and sorting in specific facilities. LDPE waste from different sectors (e.g. household packaging, commercial waste, production waste, etc.) was assumed to be used as a feedstock, to better reflect reality, where a mix of different waste streams is likely used, and not only agricultural waste. The collection and sorting processes were modelled as already described in Section 4.4.5.2 and 4.4.5.3 for the Beverage Bottle case study, and refer to collection and transport of separately collected plastic waste at the municipal level (in the absence of specific data for collection pathways related to each possible waste stream), as well as to sorting of mixed plastic waste in dedicated facilities. Both processes were implemented in the lifecycle model according to the Circular Footprint Formula (CFF), which is the default approach to handle recycling situations in the Product Environmental Footprint (PEF) context. According to this approach, only 50% of the burdens from collection, transport, sorting and recycling were assigned to the recycled material input to the mulching film system ( $A=0.5$  for LDPE used in unspecified applications), the rest being assigned to the system providing material for recycling. However, the recycled polymer input was assigned a share of the burdens associated with the supply of virgin fossil-based feedstock used in the production of the replaced virgin polymer. These were modelled as described in Section 7.4.1.1 for fossil-based LDPE, considering a share equal to 37.5% (as discussed in Section 7.4.2.1 for the Polymer Production Stage).

#### **7.4.1.3 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 for PLA production, and of European starch crops (maize, wheat, potatoes) used as a feedstock for Thermoplastic Starch, was modelled through available

EF-compliant datasets. These are based on the approach used to develop agricultural inventories in the Agri-footprint database, relying on 5-year average yield data from FAOStat (2010-2014). Activities covered in the datasets are those of seeding and seed production, fertilizer and pesticide production and application, capital goods, as well as energy use and transport for field management practices. 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 US or EU diesel mix, depending on the country of production).

#### **7.4.2 Polymer Production Stage**

The Polymer Production stage covers the activities of feedstock processing into relevant intermediates and monomer(s), the polymerisation 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 (7.4.2.1 – 7.4.2.4) describe how these activities have been modelled in the present case study.

##### **7.4.2.1 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 an aggregated, gate-to-gate dataset provided by Thinkstep. Inputs include a combination of naphtha and natural gas. All conversion processes are assumed to take place in Europe, so that the dataset reflects EU average background conditions in terms of e.g. energy generation, material supply and transport. The main conversion process involved in the supply chain is steam cracking of naphtha and natural gas, delivering the monomer Ethylene, along with Propylene, Butadiene and other relevant intermediate flows (e.g. pyrolysis gas including mixtures of Benzene, Toluene and Xylenes). The inventory of this process is mainly based on industry data, completed, where necessary, by literature data. 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 co-products. The final polymerisation process of LDPE is mainly modelled based on industry data from internationally adopted production processes, integrated with literature data when needed. No allocation is performed for this stage, being LDPE the only output from the respective production process.

##### **7.4.2.2 Recycled LDPE**

The production of recycled LDPE granulate out of sorted, post-consumer LDPE waste was approximated through 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". 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. While the modelled process is more likely representative of the recycling of rigid or semi-rigid plastic products (e.g. trays or bottles), it was considered a suitable approximation also for the recycling of flexible products (where most of LDPE waste comes from), since it is expected 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). 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).

According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the recycling process were allocated to the recycled content in LDPE mulching film ( $A = 0.5$  for LDPE used in

unspecified applications). However, the recycled content carries a share of the 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 life cycle providing the recycled material). Assuming a value of the  $Q_s/Q_p$  factor equal to 0.75 (similarly to the value recommended for LDPE used in packaging film applications, in the absence of more specific data), the allocated share of virgin LDPE production burdens is equal to 37.5% ( $A \times Q_s/Q_p = 0.5 \times 0.75 = 0.375$ ). Virgin polymer production burdens were modelled as described above (Section 7.4.2.1) for conventional, fossil-based LDPE, and in Section 7.4.1.1 for the respective Feedstock Supply.

### 7.4.2.3 Bio-based polymers

#### 7.4.2.3.1 Starch (TPS)/PBAT blend

Thermoplastic starch 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 methacrylate<sup>121</sup>. 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 reported for the corresponding processes in the Agri-footprint database (v 4.0), combined with background EF-compliant datasets for individual inputs and outputs. In these inventories, the allocation of process burdens to the different co-products (e.g. from corn wet milling) is based on the respective economic value, consistently with the fully vertically aggregated EF datasets available for starch production via wet milling (which are developed by the same data provider). Maize starch-related 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 approximated with data related to the production of Glycerine, derived from an aggregated, cradle-to-gate EF dataset. Conversely, the inventory of Glucose production was based on the *ecoinvent* dataset "[RER] Glucose production", which refers to pure Glucose produced via enzymatic hydrolysis of maize starch. However, in the implementation in the model, the maize starch input was replaced with the EU-average mix of starches from European starch crops (consistently with starch used for the main polymer). 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 (for steam). For Glycidyl Methacrylate, an average inventory related to the production of an unspecified organic chemical was applied (i.e. the *ecoinvent* dataset "[GLO] market for chemical, organic"<sup>122</sup>, in the absence of more specific data on the relevant substance.

Compounding of starch and additives to form TPS was assumed to take place in the same facility where the final starch-based polymer is produced by blending TPS and PBAT granulates. This preliminary compounding activity was not modelled separately, since the respective burdens are taken into account in the dataset used to model the final compounding step (which is described below).

<sup>121</sup> Based on information provided in the documentation of the GaBi dataset related to Thermoplastic Starch production.

<sup>122</sup> 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 remaining chemicals.

PBAT is a copolymer of 1,4-Butanediol, Adipic Acid, and Purified Terephthalic Acid (PTA). The respective synthesis (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* database. Inputs of PET precursors were replaced with the three PBAT co-monomers, considering the specific consumption reported in Schrijvers et al. (2014). These are based on a 90% synthesis efficiency, and are equal to 0.41 kg for Butanediol, 0.37 kg for Adipic Acid, and 0.33 kg for PTA (all expressed per kg of PBAT). Cradle-to-gate production inventories for these co-monomers were derived from aggregated GaBi datasets, which in the case of Butanediol and Adipic Acid refer to German background conditions, in the absence of representative data for EU conditions. Background energy inputs were combined with relevant EF datasets, while for material inputs and outputs the original *ecoinvent* datasets were kept. Infrastructure processes related to chemical factory construction were removed, to improve reliability of LCIA results in the Ozone Depletion impact category. Transport of PBAT co-monomers to the polymerisation plant was also included in the dataset, considering the default transport scenario defined in the EF context (and in the present method) for transferring of goods from suppliers to factories/users located in the EU. 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.

The inventory of the final compounding step of TPS and PBAT (including blending of raw materials, cutting, drying and cooling) was based on the *ecoinvent* dataset “[RER] Polyester-complexed starch biopolymer production”, which relies on calculations and extrapolations from highly aggregated background data from the environmental product declaration of a starch-based polymer (MaterBi). Compared to the original dataset, inputs of naphtha and natural gas were removed, as referring to the supply of raw materials and process energy associated with the production of the unspecified fossil-based copolyester used in the original blend, which is here represented by PBAT and was modelled separately (as described above). Similarly, the default input of maize starch was replaced with the average mix of starches from relevant European crops considered in this study (Section 7.1). Input quantities of starch, PBAT, and starch additives were set or adjusted to reflect the respective share in the final polymer blend (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 and waste flows reported in the original inventory were combined with EF background datasets for the modelling of the respective burdens. Finally, infrastructure processes related to chemical factory construction were removed, to improve reliability of LCIA results in the Ozone Depletion impact category.

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

#### 7.4.2.3.2 PLA/PBAT blend

PLA production from US Maize was modelled based on an aggregated, gate-to-gate inventory provided by Thinkstep. The inventory includes the processes of maize wet milling for starch production, starch hydrolysis to glucose, its fermentation to lactic acid, oligomerization of lactic acid to lactide monomer, and final polymerisation of the latter to PLA. For the maize wet milling process, economic allocation is applied to the different co-products, while for downstream conversion processes no allocation is required. The overall inventory is mainly based on industry data from internationally adopted production processes, and is completed, where necessary, by literature data. The dataset reflects European background conditions for energy and material inputs, despite the analysed mulching film scenario assumes PLA production taking place in the US and

subsequent transport of polymer resin to Europe. This is acknowledged as a limitation of this study.

Modelling of PLA transport to Europe for compounding was made according to the default scenario specified in the PEF context (and in the present method) for import of goods from suppliers located outside Europe. A transoceanic ship transport was thus assumed as main route, considering a case-specific distance of 6,000 km (for the route New York – Rotterdam)<sup>123</sup>. In addition, a default distance of 1000 km was assumed to be covered by lorry (total weight >32 t; Euro 4) for road transport of the polymer resin to and from the harbour in the US or EU.

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 route (i.e. esterification; Schrijvers et al., 2014). Polymerisation data were derived from the most recent implementation of the PlasticsEurope ecoprofile in the *ecoinvent* database (CPME, 2017), and were combined with GaBi or EF datasets for the production of the three PBAT co-monomers (1,4-Butanediol, Adipic Acid and Terephthalic Acid), as better detailed in Section 7.4.2.3.1. PBAT granules were then assumed to be transported to the compounding facility based on the default scenario specified in the PEF context (and in the Method) for transport of goods from suppliers to factories/users 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 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 polymeric blends, as reported in the *ecoinvent* dataset "[RER] Polyester-complexed starch biopolymer production". Only flows related to energy inputs, waste generation and direct emissions were maintained from the original inventory, and combined with EF background datasets for the modelling of the respective burdens. Inputs related to raw material supply (i.e. starch, naphtha and natural gas) were instead replaced with relevant raw material inputs (i.e. PLA and PBAT), modelled as described above and reflecting the share of 45% PLA and 55% PBAT in the final polymer blend. A 100% conversion efficiency was assumed for the process, considering that material losses during compounding of starch-based polymer blends were reported to be negligible (Broeren et al., 2017). Infrastructure processes related to chemical factory construction were removed, to improve reliability of LCIA results in the Ozone Depletion impact category.

#### **7.4.2.4 Transport of polymer granulate to the article production site**

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

- (i) 130 km by lorry (total weight >32 t; Euro 4);
- (ii) 240 km by train (average freight); and
- (iii) 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 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

<sup>123</sup> Defined based on the calculation tool provided by SeaRates.com, and available at <https://www.searates.com/services/distances-time>

SeaRates.com)<sup>124</sup>. Countries contributing to at least 90% of the overall import were considered in the calculation, for an overall distance equal to 6,528 km for fossil-based LDPE (see Table 7.4). Oceanic ship transport was complemented with road transport to and from the harbour in both the exporting and importing country (i.e. the EU). Road transport is made by lorry (total weight >32 t; Euro 4) along an overall default distance of 1000 km.

A similar transport scenario was assumed also for PLA (entirely imported from the US), considering an estimated overall sea distance equal to 6,000 km (for the route New York – Rotterdam). Also in this case, a 1000 km road transport by lorry (>32 t; Euro 4) was assumed for the route to and from the harbour in the exporting country (US) and in Europe.

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

**Table 7.4.** Calculation of the overall average sea distance for imports of fossil-based LDPE to Europe.

Exporting country	Import ( <sup>1</sup> ) (%)	Import (% cum.)	Distance ( <sup>2</sup> ) (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 JERICOHO- >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	-	-
Overall weighted distance				6,528

(<sup>1</sup>) Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

(<sup>2</sup>) 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 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.

<sup>124</sup> Available at: <https://www.searates.com/services/distances-time/>



### 7.4.3 Article Production Stage

Regardless of the feedstock used, mulching film was assumed to be manufactured via blown film extrusion, which is the most common method for large scale production of plastic films (Crawford and Martin, 2020). In blown film extrusion, melted plastic granules are passed through an annular die to form a thin tube, which is inflated with air from the 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 bag production (Crawford and Martin, 2020).

The inventory of the film extrusion process was derived from the aggregated, EF dataset “[EU-28+EFTA] Film Extrusion (blowing); plastic extrusion | production mix, at plant | for PP, PE, PVC, PET and PS”, which accounts for a 99% conversion efficiency. For LDPE and R-LDPE mulching film, process losses (e.g. flawed film) 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 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%). 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 LDPE film, in the absence of more specific data for recycling of pre-consumer, industrial scraps. As for starch-based and PLA-based mulching film, process losses were assumed to be incinerated, as recycling of these polymer blends is not established. For further detail on the modelling of recycling and avoided virgin material production (including the implementation of the Circular Footprint Formula), as well as of material incineration, the reader is referred to Section 7.4.5 on End of Life modelling.

### 7.4.4 Distribution Stage

The transport of mulching film from the manufacturing site to the final user was modelled based on the default transport scenario specified in the EF context and in this method for the route *factory → distribution centres → final client*. The following routes were thus considered:

- (i) 1200 km by articulated lorry (total weight >32 t; Euro 4) from factory to distribution centres (100% local supply chain); and
- (ii) 250 km round trip by van (lorry <7.5t, EURO 3; utilisation ratio of 20%) for the roundtrip from distribution centres to final users (100% local supply chain).

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

### 7.4.5 End of Life Stage

This section describes the modelling of the End of Life stage of mulching film. In particular, Section 7.4.5.1 focuses on the definition of the EU-average End of Life scenario, which is considered as a base case for the calculation of the potential impacts of the compared LCA scenarios. The remaining sections address the modelling of (non-biodegradable) mulching film collection and transport, and of the different End of Life options explored in the study. Finally, Section 7.4.5.7 provides case-specific details on the estimate of the potential contribution of mulching film to macro-plastic formation at End of Life (including product litter) and micro-plastic generation throughout the supply chain.

#### 7.4.5.1 End of Life scenario

The End of Life options available for mulching film depend on the nature of the polymer it is made of. For film made of non-biodegradable polymers (i.e. both virgin and recycled LDPE), mechanical recycling, incineration and landfilling were all considered to be viable

and currently applied (at different rates) after removal of film from land. They were thus all included in the EU-average End of Life scenario considered as a base-case

In general terms, incineration can be currently considered the most viable option for non-biodegradable mulching film, because its contamination with soil and possible agrochemicals poses a relevant issue towards both mechanical recycling and landfilling of the film itself. Indeed, despite not being an option to be pursued, landfilling of soil contaminated mulching film can become extremely costly because of the increased volume of material to be disposed of (Steinmetz et al., 2016). On the other hand, while being a potentially desirable alternative to landfilling, mechanical recycling of used mulching film is reported to be only feasible for low levels of (soil) contamination (i.e. less than 5% by weight), with the current average contamination level being much higher (40% - 50%; Steinmetz et al., 2016). In Europe, only Germany and Ireland recycle a consistent amount of used mulching film, thanks to an advanced collection system (PRE, 2017).

Apart from these general considerations, no sufficiently representative data are available on the rates in which the mentioned options are currently applied to mulching film in Europe. Therefore, the EU-average recycling rate estimated for flexible LDPE/LLDPE products by Plastics Recyclers Europe was applied (i.e. 20%; PRE, 2019), while assuming that the recycling process is operated in a way that allows handling of post-consumer film with non-negligible level of contamination by soil. Moreover, the average incineration and landfilling rates estimated for total plastic waste at the EU level were considered, i.e. 39% and 30% (EC, 2018a), which in relative terms are equal to 56% for incineration and 44% for landfilling. The share of non-recycled mulching film (70%) was thus split among these two options according to the mentioned proportions, corresponding to an overall EU-average End of Life scenario including 20% mechanical recycling, 45% incineration and 35% landfilling.

As reported above, the application of all these End of Life options requires mulching film to be collected from the field after use, and subsequently transported to the intended treatment. However, at present collection is normally not capable of ensuring a complete removal of film from the field, while a collection rate (efficiency) depending on the film thickness typically applies: the thicker the film, the higher the collection rate. Thinner LDPE films are easily damaged by sunlight and other weathering agents, resulting in a partial collection and increasing the amount of residual film left on the soil (Steinmetz et al., 2016). However, the relation between collection rate and thickness is not clearly defined. Plastic Recyclers Europe assumes a 60% collection rate for mulching film with a thickness of 25  $\mu\text{m}$ . Conversely, Organic Waste System estimates that up to 68% of the material may remain uncollected in soil for a 10  $\mu\text{m}$  thick mulching film (i.e. 32% collection rate; OWS, 2017), with the collection rate increasing up to 90% for mulching film thicker than 25  $\mu\text{m}$ .

In this study, a thick (35  $\mu\text{m}$ ) LDPE mulching film was considered (Section 7.2), which is higher than the minimum thickness required by the standard EN 13655 for mulching film intended to be removed from soil after use (i.e. 20  $\mu\text{m}$ ). Therefore a 90% collection rate was assumed, according to the largest value reported above for film with the most similar (but still lower) thickness (i.e. 25  $\mu\text{m}$ ). The shares of the different End of Life options considered for non-biodegradable mulching film were thus applied to 90% of the applied film, the rest being left in the field. In this respect, it is noted that negative effects on soil quality and resulting crop growth (e.g. yield) due to the accumulation in soil of uncollected non-biodegradable mulching film have been reported, especially for cotton crop cultivation in China (Müller & Müller, 2015). In the long term, a reduced yield may generate additional (indirect) environmental impacts due to an increased demand of agricultural inputs (e.g. water and fertilisers) to keep comparable production levels. However, no quantitative evidences are currently available in the literature for horticulture in the EU, and it was hence not possible to quantitatively assess the potential impact of this effect (as part of the additional environmental information). On the other hand, a sound handling of used mulching film is expected to occur in Europe, with lower

residues being left on the field compared to e.g. China (as also reflected by the high collection rate assumed in this study, i.e. 90% as discussed above). Therefore, effects on soil quality and yield can reasonably expected to be more limited.

Residual fragments of non-collected mulching film were here modelled as direct (macro-plastic) release to soil, and quantified in the corresponding inventory-level indicator (see Section 7.4.5.7). No further burdens after this release were modelled, such as any additional release of additives or micro-plastics from further (possible) fragmentation or degradation (similarly to the other plastic articles investigated in this project once littered into the environment). This is due to the lack of data on the use of additives in mulching film production (see Section 7.3) and incomplete understanding of the fate of plastic products or pieces once released into the environment (including degradation and fragmentation pathways).

As for biodegradable mulching film (starch-based and PLA-based), in-situ biodegradation was the unique End of Life option explored, assuming that all material components (polymer, additives) are suitable for biodegradation (although this has not yet been completely proven).

#### **7.4.5.2 Modelling of mulching film collection from the field and transport to treatment**

As reported in Section 7.4.5.1, non-biodegradable mulching film needs to be removed from the field at the end of the growing season, before being routed to the different End of Life options (mechanical recycling, incineration or landfilling). Moreover, considering the thickness assumed for both virgin and recycled LDPE films (35 µm) a collection rate of 90% was applied (see Section 7.4.5.1 for further discussion on this assumption).

As an approximation of the removal process, the GaBi dataset “[GLO] Soil cultivation; stubble cleaning (medium, 67 kW)” was applied in the modelling, in the absence of more specific data for mulching film (and of EF datasets for soil treatment operations). Also the amount of soil contaminating the mulching film after use was considered to be collected, and quantified as 45% of the removed film by weight (average of the range reported by Steinmetz et al. 2016, i.e. 40-50%). However, contamination by soil does not affect the burdens from the removal process, these being proportional to the area of land to be covered and not to the mass of material to be collected.

Transport of collected film and contaminating soil to the waste treatment facilities was assumed to take place along an overall average distance of 100 km, and was modelled based on the EF-compliant dataset “[EU-28+3] Articulated lorry, Total weight 28-32 t, mix Euro 0-5”. The current average European diesel mix was considered as an input to this process.

#### **7.4.5.3 Modelling of sorting and recycling**

Before recycling, contaminated mulching film collected from the field was assumed to be firstly sorted in specific facilities, similarly to plastic waste from municipal collection. 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.

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 data collected from different dual-stream and single-stream sorting facilities in the Country). While the latter are more representative of sorting of smaller plastic articles such packaging items or other small-sized consumer goods from municipal collection, they were considered a reasonable approximation for the burdens of any specific sorting operation mulching film may undergo (in the absence of more representative data for agricultural plastics). For implementation in the model, input/output data reported in the

mentioned source were complemented with background EF datasets representative of EU-average conditions, as better detailed in Section 4.4.5.3 (Table 4.9). A 100% sorting efficiency was assumed for waste mulching film, which entirely consists of one of the targeted materials for recycling. Moreover, although a portion of contaminating soil may be unintentionally removed during sorting, it was conservatively assumed that no losses occur at this stage, and the entire soil content is transferred, along with sorted film, to the next recycling step. The presence of contaminating soil was hence addressed by modelling the sorting process (whose burdens are proportional to the mass of input material) considering the contribution of both the film and of the respective contamination. In other words, the overall burdens of the sorting process were increased proportionally to the amount of soil entering the treatment. In the modelled process, burdens from sorting are those associated with the supply of the material and energy inputs required to carry out this waste treatment activity.

For LDPE-based film recycling, no specific inventory data are available. Therefore, this process was modelled through an aggregated, EF-compliant 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)<sup>125</sup>. The latter is developed based on literature data for these unit operations, 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).

While the dataset is likely more 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. On the other hand, the applied inventory does not account for the burdens of any specific additional process step that may be performed for the removal of contaminating soil. Therefore, it was here assumed that soil is entirely removed during the conventional washing/flotation steps already included in the process, and routed to wastewater leaving the plant for further treatment. Since the burdens from wastewater treatment are already accounted in the aggregated dataset, the contribution of soil to such burdens (mostly due to an increased sludge generation from the initial settling stages) were not separately modelled. However, similarly to the sorting process described above, the burdens due to the presence of soil were accounted considering that an increased mass of material (i.e. both mulching film and soil) enter the recycling process, thus increasing the respective burdens proportionally to the amount of soil collected with the film. The applied dataset is developed based on literature data for these unit operations, and accounts for an overall recycling efficiency equal to 84% (on the input material). Process waste and scrap are sent to incineration, consistently with the typical fate of plastic recycling residues, and the assumption performed in the PP recycling process.

Recycled polymer granulate was assumed to replace virgin granulate of the same material, whose primary production burdens were credited to the system. Both fossil-based and bio-based LDPE were assumed to be replaced, according to the current average mix of the two production routes. However, the estimated share of the bio-based pathway is currently marginal, being equal to only 0.2% for LDPE (which is for 99.8% fossil-based)<sup>126</sup>. To account for the lower overall quality of the recycled polymer compared to the replaced virgin material, a substitution ratio equal to 0.75 was considered for LDPE, according to default values specified in the PEF context for LDPE derived from

<sup>125</sup> [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.

<sup>126</sup> The share of bio-based LDPE consumed in Europe was estimated based on the global production capacity of bio-based PE in 2018 (European Bioplastic, 2019; 200 kt) and of PE as a whole in 2016 (PlasticsInsight, 2019; 103 Mt).

(packaging) film recycling. To model the burdens of avoided primary production, the same datasets (or combination of datasets) used for the modelling of upstream production of the two polymers were applied (as described in Sections 4.4.2, and 4.4.1 for the related feedstock<sup>127</sup>). This was made for consistency reasons, and to avoid possible distortions by applying different datasets from other sources.

According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the sorting and recycling processes were allocated to the system ( $A = 0.5$  for PE used in unspecified applications). Similarly, only 50% of the benefits from avoided virgin material production were assigned to the system itself.

#### **7.4.5.4 Modelling of incineration**

For conventional fossil-based polymers, including LDPE, aggregated, material-specific incineration datasets are available from the EF database, and were applied to model the treatment of mulching film made of both virgin and recycled LDPE in a municipal waste incineration plant. Similarly, for starch and PLA used as copolymers in the respective polymer blends, partially aggregated, material-specific inventories from the GaBi database were applied (no EF datasets are available for these polymers).

All the selected datasets are developed based on a waste-specific incineration model, described in detail in Section 4.4.5.4. 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 general approach to handle energy recovery situations adopted in the EF context and in the method (Report I), the product system generating the waste material sent to incineration (i.e. the mulching film life cycle, in this case) takes the full burdens from the incineration process. However, it is credited for 100% of the benefits from avoided production of conventional energy (electricity and heat) replaced by energy recovered from waste. While in EF incineration datasets these credits are already accounted for in the aggregated inventory, for GaBi datasets they were added to the main process inventory. Therefore, the EU residual electricity grid mix (as modelled in the EF 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 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 includes 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. Also, thermal energy from waste (11%) was excluded, as according to the approach described above for the handling of energy recovery situations, 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 ready-to-use incineration datasets were available. A disaggregated, material-specific inventory was thus developed, based on the most recent version of the calculation tool developed by Doka (2009a) for the modelling of material incineration within municipal solid waste incineration plants. The model operates similarly to the one used for the

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<sup>127</sup> For Bio-LDPE, please refer to Sections 5.4.2.1 and 5.4.1.1 of the Packaging Film case study.

development of the EF incineration datasets as 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 tool also allows to adjust other technological parameters to the relevant geography or scope, including energy efficiencies, the share of alternative NO<sub>x</sub> control technologies, and a few other specific parameters. For more details on how these parameters were set, and on the assumed PBAT composition and energy content, the reader is referred to Section 5.4.5.4 related to the packaging film case study. Credits associated with recovered energy were calculated as described above for existing GaBi datasets (i.e. electricity from the EU-average 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, background EF datasets were applied.

Incineration of soil contaminating used mulching film collected from the field was not modelled, due to the lack of representative data on the average soil composition at the EU level to be used for modelling purposes. However, this aspect may be improved in the future.

#### **7.4.5.5 Modelling of landfilling**

Landfilling of mulching film made of conventional, non-biodegradable polymers (i.e. LDPE and R-LDPE) was modelled based on a common, aggregated EF dataset representing disposal of (fossil-based) plastic waste in a managed, municipal solid waste landfill (*[EU-28+EFTA] Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific sites)*). The underlying inventory is material-specific, but refers to the average chemical composition and degradability values of generic plastic waste, rather than to those of the specific polymer. 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<sup>128</sup>. 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 emissions to the environment over a 100 year time horizon. Emissions occurring beyond 100 years are not accounted for in the model. Landfill gas generation is calculated based on the organic carbon content in the waste and the respective degradation rate assumed over 100 years (not reported), while, 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 rate of landfill gas, energy efficiencies of engines, collection rate of leachate and respective treatment efficiencies). Further details on how these parameters were set to reflect the EU-average situation are provided in Section 4.4.5.5.

Landfilling of soil contaminating used mulching film collected from the field was not modelled, due to the lack of representative data on the average soil composition at the EU level to be used for modelling purposes. However, this aspect may be improved in the future.

#### **7.4.5.6 Modelling of in-situ biodegradation**

According to the recommendations reported in the Method (Report I, Section 4.4.12.8, in-situ biodegradation of starch-based and PLA-based mulching film was modelled by

<sup>128</sup> While more polymer-specific landfilling datasets are available from other databases (i.e. *ecoinvent*), the mentioned EF dataset for landfilling of generic plastic was selected, as specifically referring to EU as the reference geography (in contrast to available polymer-specific datasets), and to keep consistency with the overall dataset selection “hierarchy” followed in this project (Section 3.5).

1 assuming an overall biodegradation rate in soil equal to 90% over 100 years (in line with  
2 the minimum biodegradation percentage required by EN 17033). While data from real  
3 testing are reported in the literature (UBA, 2018), these refer to films made of polymer  
4 blends with a different composition compared to those assumed in this case study (in  
5 terms or relative shares between blending copolymers) and thickness is not specified for  
6 such polymers (or it is not comparable to the values assumed here). Moreover, in most  
7 cases such values refer to shorter testing periods (3-9 months) than the maximum  
8 duration allowed in the above mentioned standard (24 months), and testing is not  
9 necessarily carried out according to the prescribed standard (i.e. EN ISO 17556). On the  
10 other hand, it must be noted that a higher (94%) biodegradation rate was observed over  
11 6 months for PLA/PBAT film (9% PLA and unspecified thickness). Conversely, much lower  
12 rates are reported for starch blends, i.e. 37% over 9 months for a TPS/PCL blend (16/75  
13 plus 9% additives), and 18% over 4 months for Mater-Bi film (both with unspecified  
14 thickness). For consistency reasons, and to avoid potentially biased comparisons, the  
15 same minimum biodegradation rate was hence assumed in this study for both polymer  
16 blends (i.e. 90% as reported above).

17 The modelled inventory included carbon emissions from biodegradation, calculated in  
18 accordance with the considered degradation rate (which indeed expresses the share of  
19 organic carbon in the polymer mineralised to CO<sub>2</sub> under testing conditions), and  
20 assuming that all mineralised carbon is aerobically converted to CO<sub>2</sub>. Fossil and biogenic  
21 CO<sub>2</sub> emissions were separately quantified, according to the origin of carbon in the  
22 different copolymers (i.e. biogenic for TPS and PLA, fossil for PBAT) and considering the  
23 material composition reported in Table 7.5. Since the latter does not include any metal  
24 that would not be degraded, no emissions to soil of these substances were modelled.  
25 Moreover, similarly to residues of non-biodegradable mulching film left on the field after  
26 collection, no release of additives or of any residual compounds from biodegradation was  
27 modelled, as it was the case of any non-degraded plastic fragments that may be released  
28 (transferred) to other environmental compartments before (full) degradation occurs. This  
29 is due to the lack of data on the use of additives in mulching film, and to incomplete  
30 understanding of real biodegradation pathways of biodegradable polymers in soil.  
31 However, from a theoretical point of view, if all polymer components are ultimately  
32 biodegradable, and no hazardous substances are included in the respective composition,  
33 at the very end of the biodegradation process only CO<sub>2</sub>, water and new soil biomass is  
34 generated (provided no relevant transfer of non-degraded plastic fragments or of any  
35 intermediate compounds occurs towards other environmental compartments during  
36 biodegradation).

37

**Table 7.5.** Elemental composition of PLA, TPS and PBAT copolymers considered for the modelling of in-situ biodegradation of biodegradable mulching films <sup>(1)</sup>.

Element	TPS <sup>(2)</sup>	PLA <sup>(3)</sup>	PBAT <sup>(4)</sup>
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	-

<sup>(1)</sup> Starch-based mulching film is composed of 40% TPS and 60% PBAT, while PLA-based film consists of 45% PLA and 55% PBAT.

<sup>(2)</sup> Based on stoichiometry, and considering that TPS consists of 75% starch and 25% plasticisers/additives (equally split among Glycerol, Sorbitol and Glycidyl Methacrylate).

<sup>(3)</sup> Based on results from composition analysis related to Ingeo polymer available at: <https://www.natureworksllc.com/What-is-Ingeo/Where-it-Goes/Incineration>

<sup>(4)</sup> Based on stoichiometry.

#### 7.4.5.7 Contribution to macro- and micro-plastic generation (including littering)

The contribution to macro- and micro-plastics loss and release of the analysed mulching films scenarios was estimated according to the framework outlined in Annex B of the Method (Report I), and summarised in Section 4.4.5.6 of this report in terms of underlying quantification methods and approaches. In this section, the case-specific parameters considered to apply the latter to the assessed mulching films LCA scenarios are reported.

##### 7.4.5.7.1 PLP method

Regarding the estimate of the loss and release of macro-plastics at the End of Life stage (from product littering and waste mismanagement), the product-specific parameters reported in Table 7.6 were considered to apply the first level of the framework (“*PLP method*”) to mulching films (derived from Peano et al., 2020). Note that 10% of non-biodegradable, fossil-based mulching films was considered to remain uncollected on soil (Section 7.4.5.1) and therefore littered, while partially bio-based mulching films are intentionally left on soil for biodegradation and they were thus considered to be totally released to the terrestrial environment. However, the applied framework excludes the assessment of biodegradation due to the lack of complete understanding of fragmentation and biodegradation pathways of biodegradable plastic products in soil and in the environment in general. Therefore, consistently with the other plastic articles investigated in this project when littered into the environment, the release refers to the product when it appears in the environment itself, taking into account its subsequent redistribution between environmental compartments (e.g. transfer from freshwater to ocean). This is different from the ultimate fate and release once biodegradation has taken place.



**Table 7.6.** Case-specific parameters of the PLP method applied to the mulching films LCA scenarios.

Parameter <sup>(1)</sup>	Value (LDPE, R-LDPE)	Value (Starch-based, PLA-based)
Littering rate (LR <sub>lit</sub> ) (%)	10	100
Release rate to ocean (Rel <sub>ocean</sub> ) (%)	5	0
Release rate to the terrestrial environment (Rel <sub>terenv</sub> ) (%)	95	100

<sup>(1)</sup> For details on the meaning of each parameter, refer to Annex B of the Method (Report I).

No case-specific parameters are required to calculate the supply-chain loss and release of micro-plastics to ocean and to the terrestrial environment by applying the *PLP method*. Indeed, the contribution of the considered micro-plastics sources (pellet loss and tire abrasion) depends on the magnitude of the relevant life cycle processes (i.e. polymer production and road transport) in the specific mulching films scenario (which in turns depend on the reference flow<sup>129</sup> of each scenario). However, the quantification is made by means of default parameters that are not affected by the type of product, polymer or feedstock source (as reported in Annex B of the Method (Report I)).

#### 7.4.5.7.2 Expanded PLP method

The parameters required to calculate the micro-plastics loss and release from marine coatings and road markings are not case-specific, and are reported in Annex B of the Method (Report I). Similar to the other considered micro-plastics sources, the contribution of these additional sources depends on the intensity of relevant (transport) processes in the specific product inventory, by means of the above-mentioned case-unspecific parameters.

As for the calculation of the additional contribution of plastic waste exported for recycling to the loss and release of macro-plastics from waste mismanagement, the case-unspecific parameters described in Annex B of the Method (Report I) were applied, in combination with the EU-average recycling rates estimated for each type of mulching films (Section 7.4.5.1), i.e. 20% for both virgin LDPE and R-LDPE. Note that biodegradable mulching films are not collected from the field after use, and therefore do follow any End of Life treatment pathway that can contribute to plastic release via waste mismanagement.

### 7.4.6 Calculation of iLUC impacts

As a base case, the contribution of Indirect Land Use Change (iLUC) to the potential Climate Change impact of bio-based mulching film alternatives was calculated according to the approach outlined in Section 4.4.16.4 of the Method (Report I).

In order to apply (recalculated) iLUC GHG emission factors from the EU 2015/1513 Directive (EC, 2015), the specific land demand for the production of relevant crops (m<sup>2</sup>\*year / kg crop) was calculated first. The calculation was based on the aggregated amount of arable and agriculture land occupation flows reported, for the relevant country (e.g. US or Europe for maize), in the dataset used to model the production of the specific crop. If the geography of such flows was not specified, all the arable and agricultural land occupation flows reported in the dataset were aggregated. These estimates were checked against values of land demand calculated based on 5-years average crop yields from FAOSTAT (FAO, 2019), which were found to be generally aligned with the former (absolute variation between 3% and 20%). Hence, the values estimated based on land

<sup>129</sup> The reference flow is the mass of mulching film material required to fulfil the functional unit.

1 occupation flows were applied, for consistency with the overall LCI modelling of  
2 scenarios.

3 Specific land demand for crop production was then converted into a demand per  
4 functional unit, based on the specific consumption of crop for polymer production (kg  
5 crop/kg polymer, consistently with the values applied in the LCI modelling) and the  
6 amount of polymer needed to fulfil the functional unit (reference flow) in the relevant  
7 scenario. The iLUC contribution to the Climate Change impact was finally calculated by  
8 applying to the latter the recalculated GHG emission factors from the Directive. The  
9 described calculation steps to estimate the iLUC contribution to the potential Climate  
10 Change impact are summarised in Table 7.7.

11

**Table 7.7.** Calculation of the iLUC contribution to the potential Climate Change impact of mulching film LCA scenarios relying on partially bio-based polymers.

Scenario/ Polymer	Feedstock	Land demand for crop production <sup>(1)</sup> [m <sup>2</sup> *y/kg <sub>crop</sub> ]	Crop demand for polymer production [kg <sub>crop</sub> /kg <sub>polymer</sub> ]	Polymer demand per F.U. [kg <sub>polymer</sub> /FU]	iLUC GHG emission factor [kg CO <sub>2</sub> eq./ (m <sup>2</sup> *y)]	iLUC climate change impact [kg CO <sub>2</sub> eq./FU]
S3 – Starch- based mulching film (TPS/PBAT)	Maize (EU)	1.34 (1.37)	0.134	92.4	0.0612	1.01
	Wheat (EU)	1.52 (1.75)	0.121	92.4	0.0612	1.04
	Potatoes (EU)	0.376 (0.301)	0.140	92.4	0.0612	0.298
	<i>Total</i>					2.35
S4 – PLA-based mulching film (PLA/PBAT)	Maize (US)	1.06 (0.939)	1.39	83.6	0.0612	7.54

(3) Based on arable and agriculture land occupation exchanges reported in the dataset applied to the modelling of the specific crop. Values in parenthesis refer to land demand calculated based on crop yield data from FAOSTAT (5-years average), and are reported as a reference.

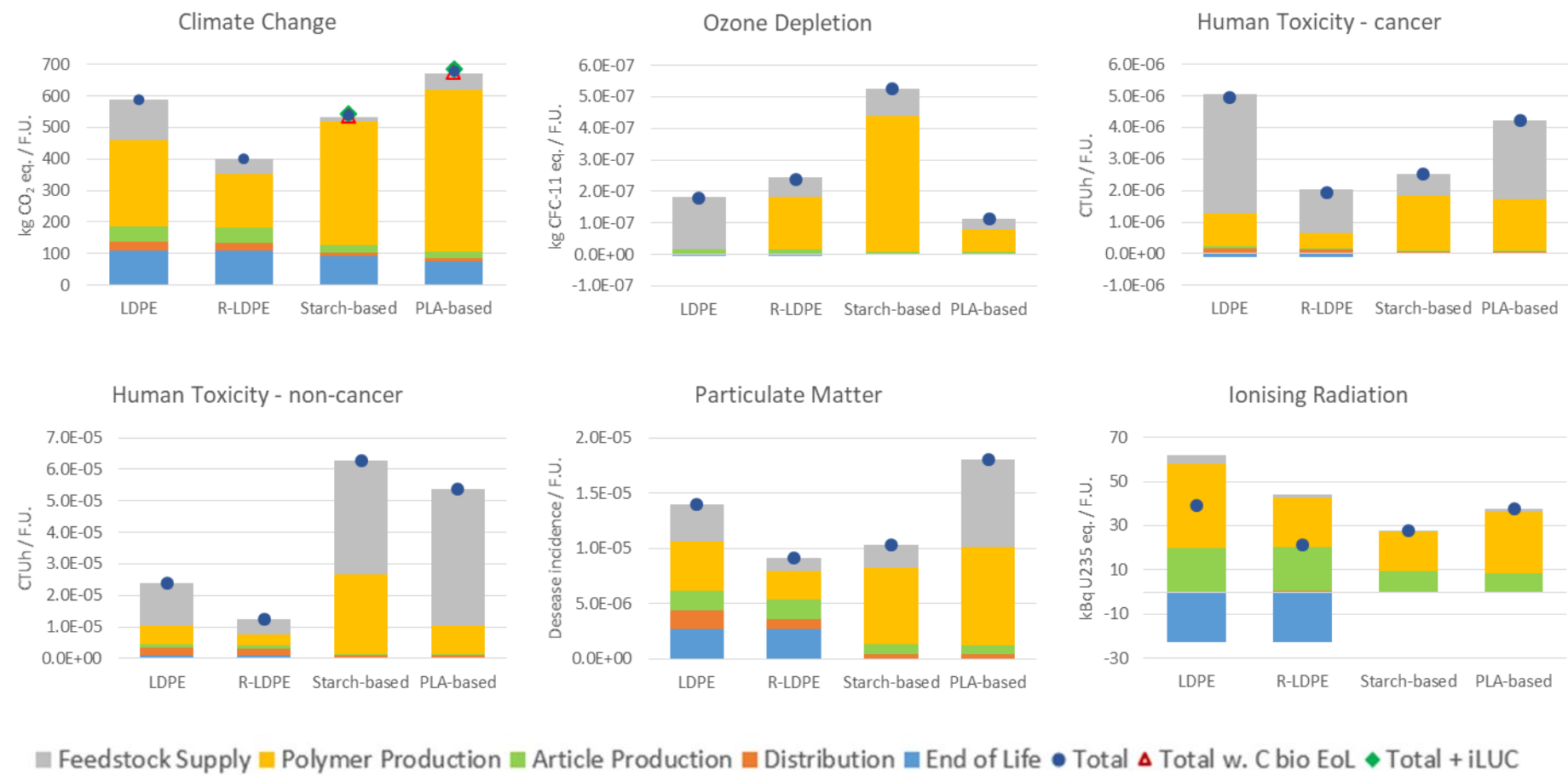
## 7.5 Life Cycle Impact Assessment results

The characterised potential impacts of the examined scenarios (with and without the iLUC contribution to Climate Change) are reported in Figures 7.5 to 7.7. These also show the breakdown of contributions from the main lifecycle stages, which include:

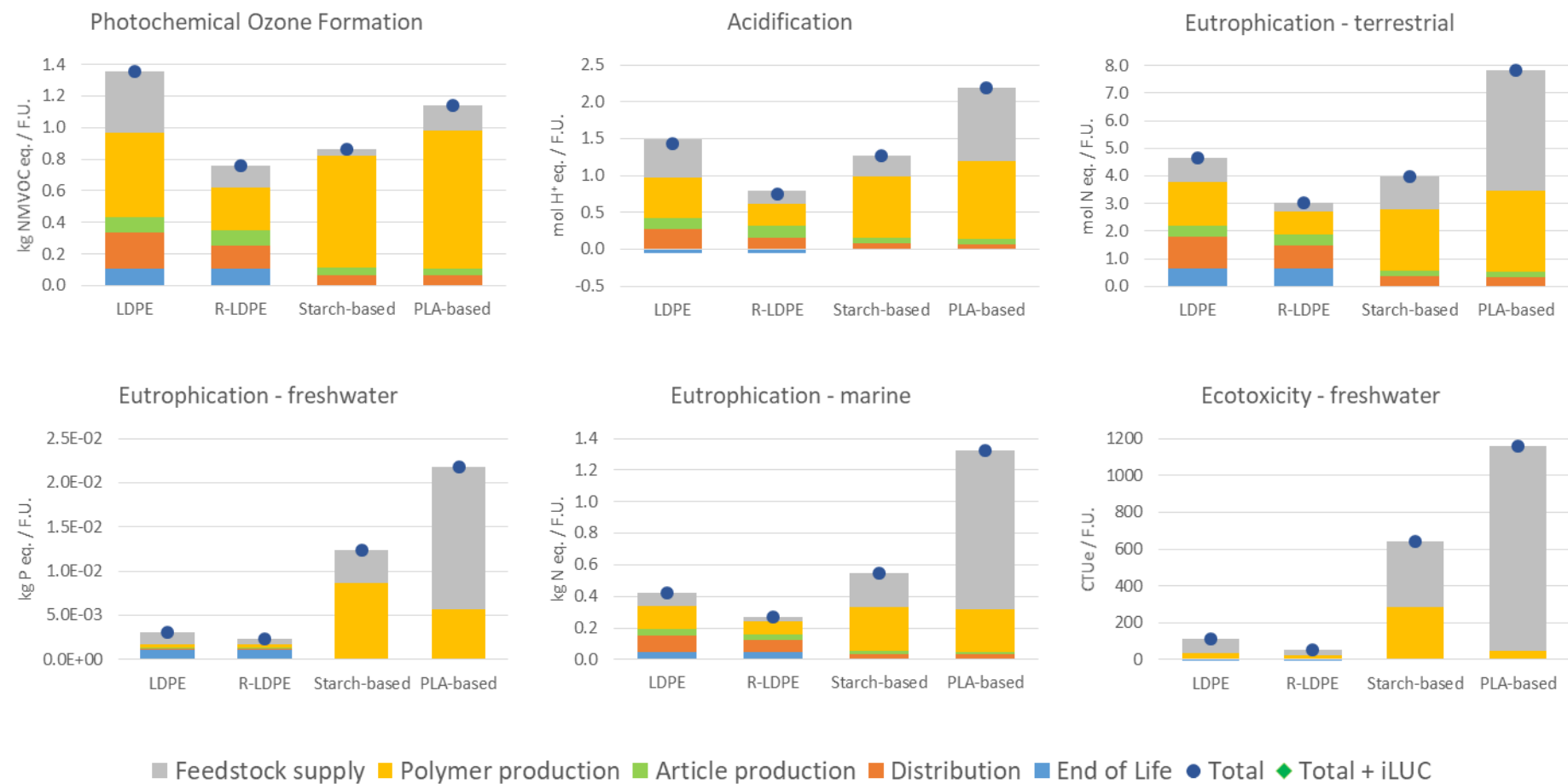
- i) *Feedstock Supply*, i.e. oil/natural gas extraction, transport and possible refining, transport of naphtha from refinery to downstream users (fossil-based polymers); collection, transport and sorting of post-consumer plastic waste (recycled polymers); or crop cultivation and transport to further processing (bio-based polymers);
- ii) *Polymer Production*, i.e. all gate-to-gate activities from feedstock processing to monomer/intermediate production and polymerisation, including any transport among these as well as transport of polymer granulate to the mulching film manufacturing site;
- iii) *Article Production*, i.e. conversion of the polymer into mulching film by blown film extrusion;
- iv) *Distribution*, i.e. transport of mulching film from the manufacturing site to the final user; and
- v) *End of Life*, i.e. mulching film collection, transport and treatment (or in-situ biodegradation), as well as any avoided processes from downstream displacement of virgin materials and energy.

Figure 7.8 shows the contribution of each sub-category to the Climate Change impact (i.e. from fossil, biogenic and land use/land transformation GHG emissions), while the overall characterised, normalised and weighted impact assessment results are available in Annex B.4, in Table B.4.5 to Table B.4.7. The latter are not reported in this section to keep readability of the document, despite this is not in alignment with the position suggested in the LCA report template provided in the Method (Report I).

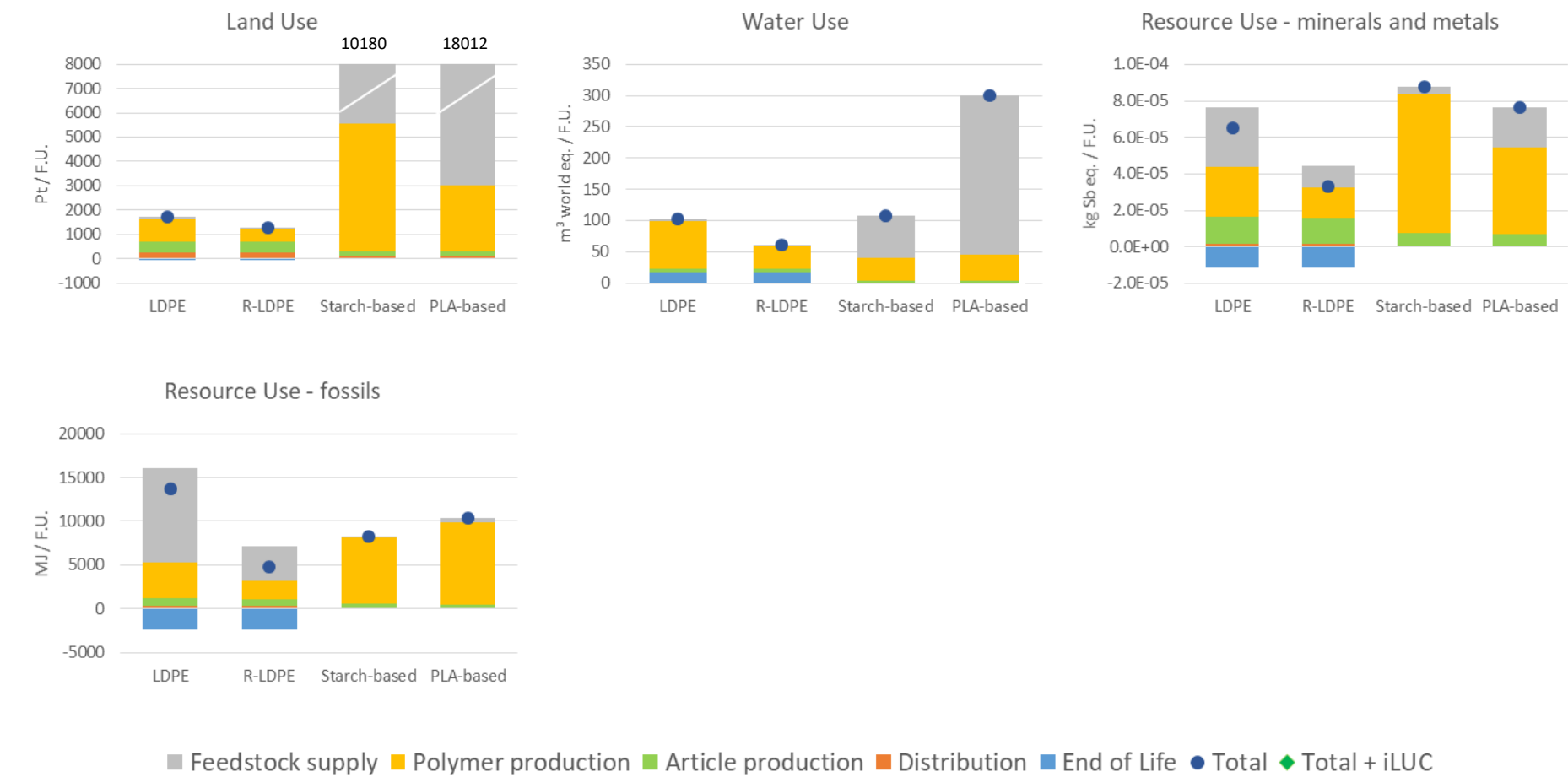
Note that scenario impacts presented in Figures 7.5 to 7.7 refer to the EU-average End of Life scenario (as described in Section 7.4.5.1), and represent the net impacts from waste management activities, resulting from the balance between real burdens and benefits (if any). Potential impacts calculated assuming 100% of used mulching film being routed to each viable End of Life option are presented in Figures 7.9 to 7.11. We warn that this analysis should not be interpreted as a direct LCA comparison of alternative End of Life management options for mulching film, because a product-based perspective is considered, and allocations among consecutive product life cycles are applied to the recycling pathway. Conversely, in a typical waste management LCA that compares alternative End of Life options (i.e. with a functional unit of 1 tonne of waste managed), the recycling pathway would be assigned the full burdens and benefits it involves, as there is no need to break the system mass flows between the system recycling and the one using the recycled feedstock (i.e. system perspective; no allocation needed).



**Figure 7.5.** Potential impact of mulching film LCA scenarios for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity - non-cancer, Particulate Matter and Ionising Radiation. Climate Change impacts denoted with "C bio EoL" account for the contribution of biogenic carbon not released after 100 years from in-situ biodegradation of bio-based mulching film

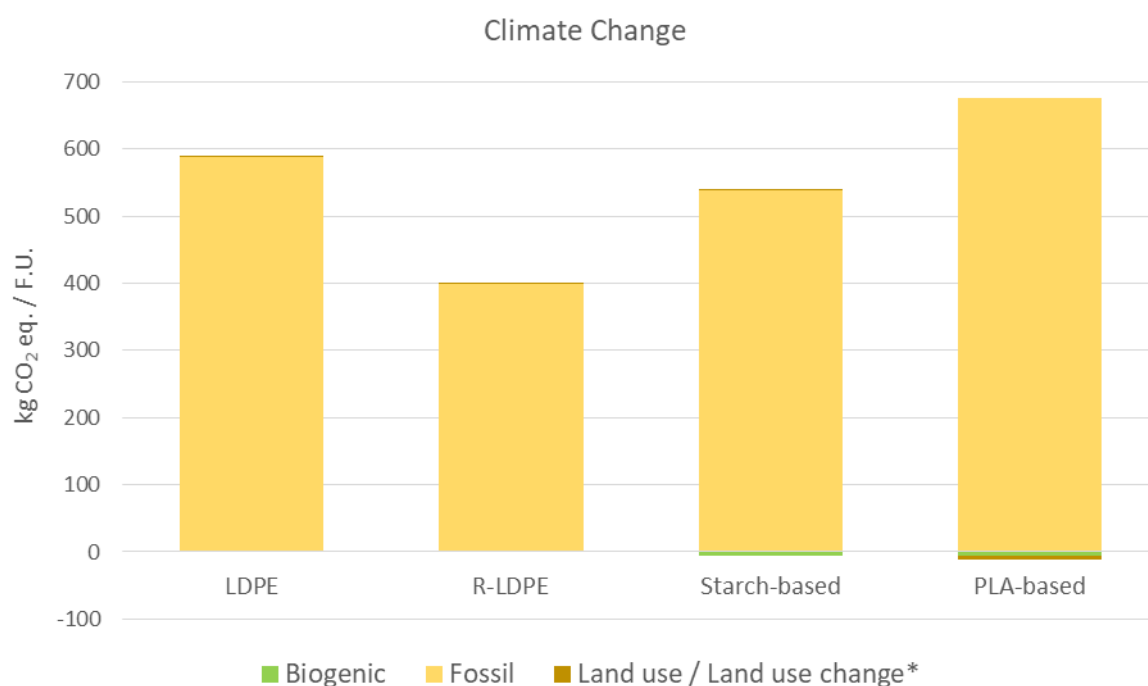


**Figure 7.6.** Potential impact of mulching film LCA scenarios for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity - freshwater.



**Figure 7.7.** Potential impact mulching film LCA scenarios for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils. Note that in some impact categories a part of the results is out of scale and is curtailed.

1

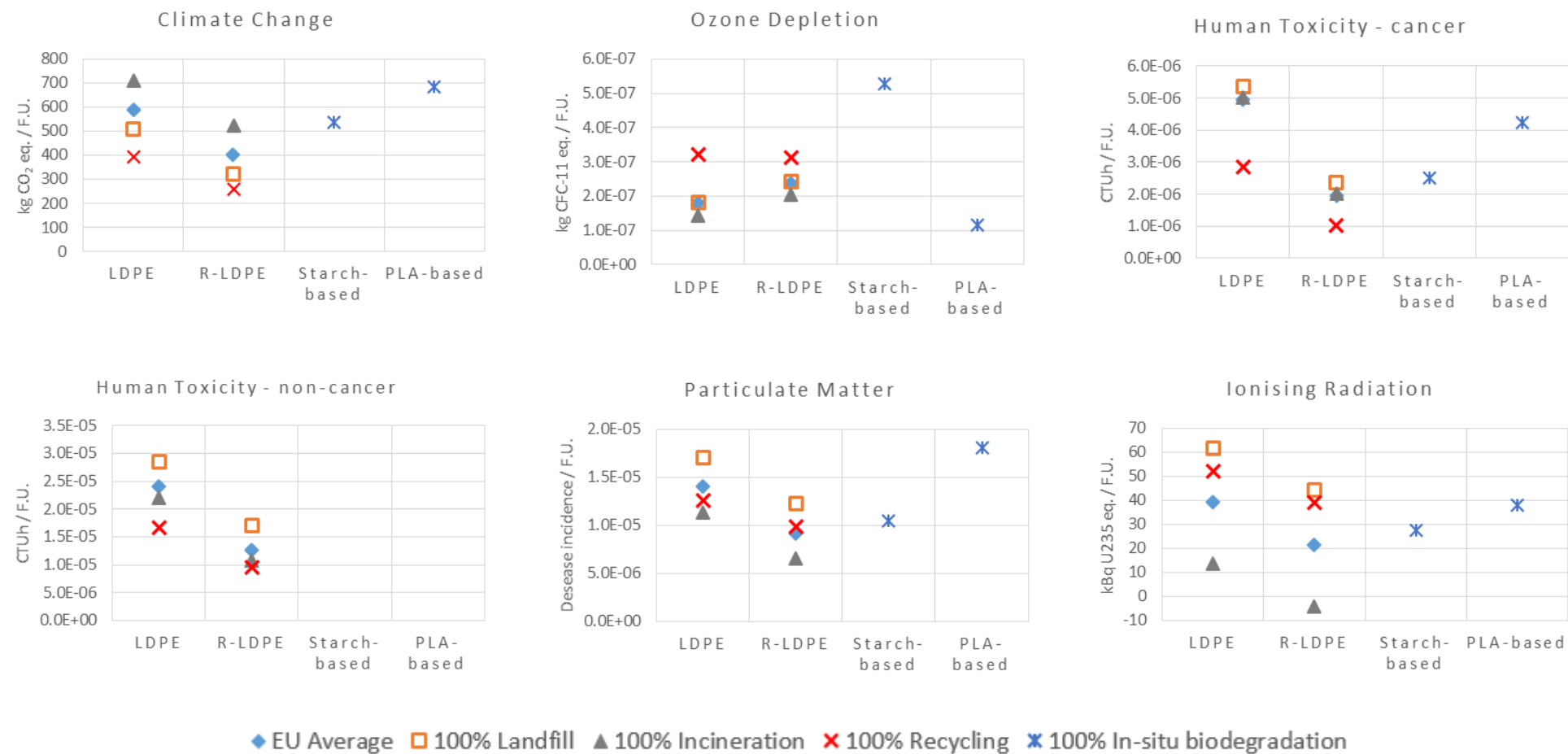


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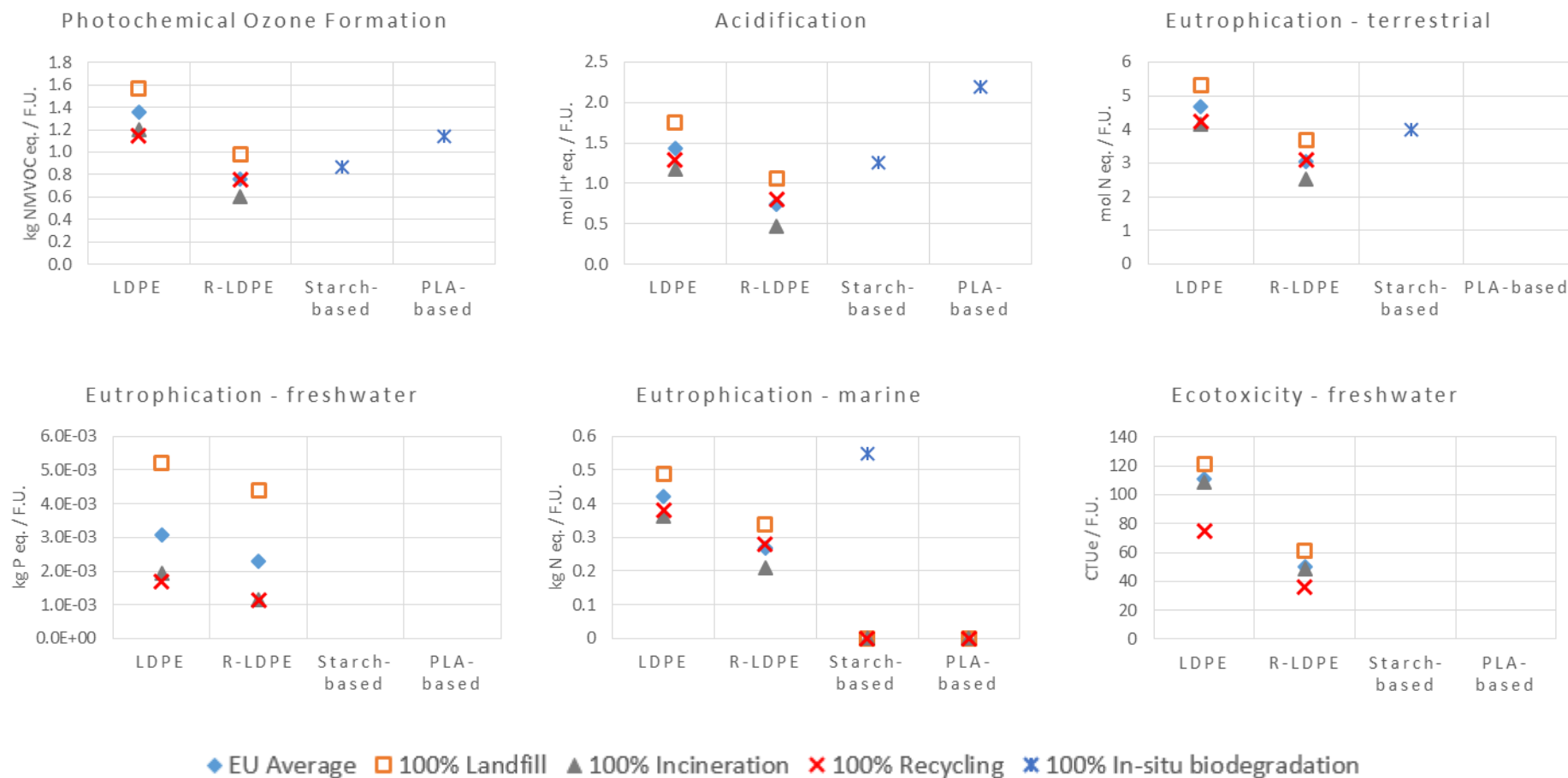
3 **Figure 7.8.** Contribution of fossil, biogenic, and land use/land transformation GHG emissions to  
4 the total Climate Change impact of mulching film LCA scenarios (note that the contribution of iLUC  
5 to land transformation emissions is excluded).

6



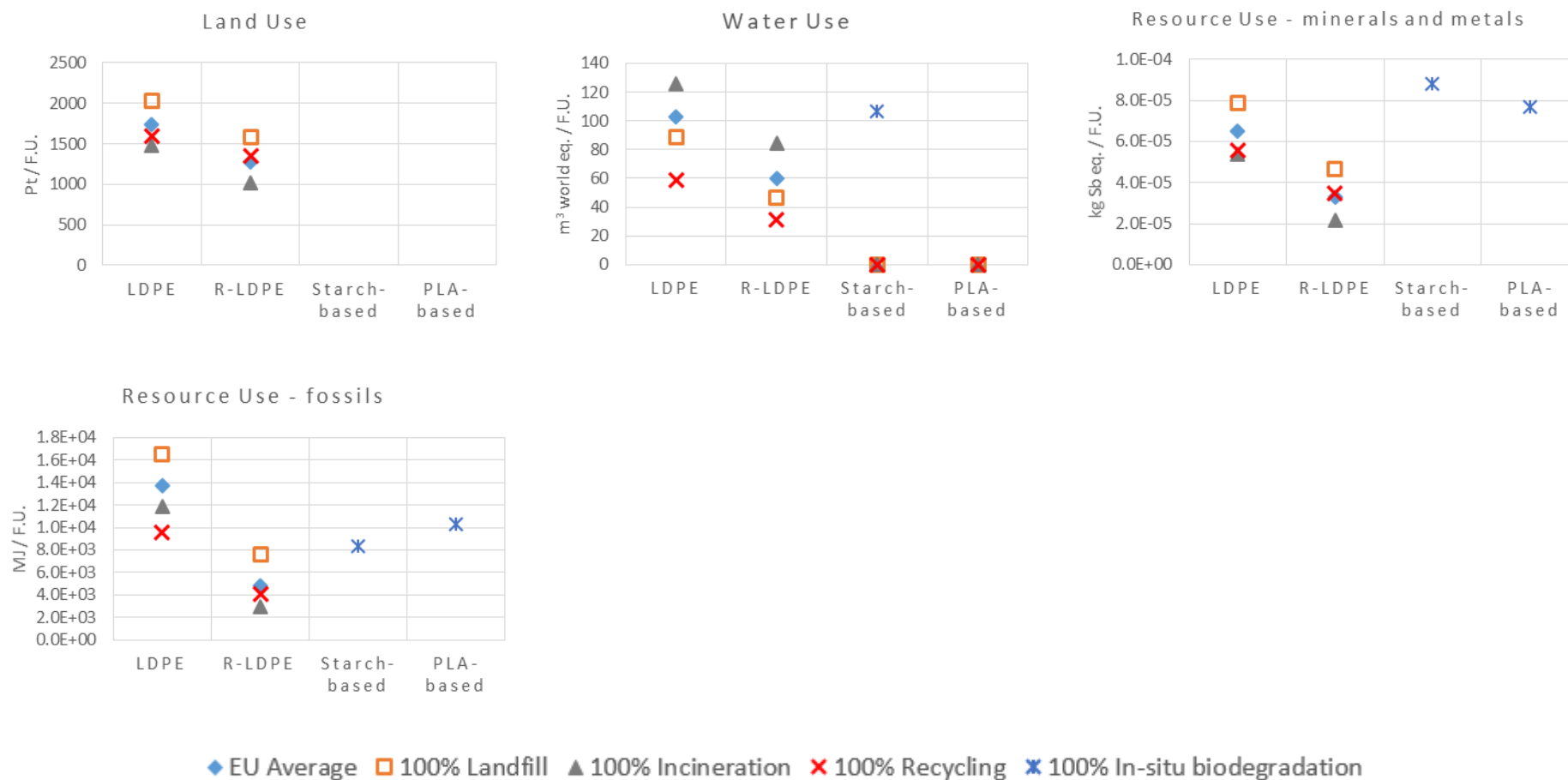


1 **Figure 7.9.** Potential impact of mulching film LCA scenarios for different End of Life options, for the categories of Climate Change, Ozone Depletion,  
2 Human Toxicity - cancer, Human Toxicity – non-cancer, Particulate Matter and Ionising Radiation. Note that in some categories the impact of in-situ  
3 biodegradation of one or both biodegradable film alternatives is out of scale and is not reported.  
4



**Figure 7.10.** Potential impact of mulching film LCA scenarios for different End of Life options, for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater. Note that in some categories the impact of in-situ biodegradation of one or both biodegradable film alternatives is out of scale and is not reported.

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**Figure 7.11.** Potential impact of mulching film LCA scenarios for different End of Life options, for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils. Note that in some categories the impact of in-situ biodegradation of one or both biodegradable film alternatives is out of scale and is not reported.

## 7.6 Additional Environmental Information

This section presents additional environmental impacts or aspects going beyond the default impact categories considered in the present method, which reflect official categories adopted in the PEF context. Additional environmental issues addressed include the impact from indirect Land Use Change (iLUC) on Climate Change, potential Biodiversity impacts, as well as the contribution of the investigated mulching film scenarios to macro-plastics formation at End of Life (including product litter) and to micro-plastics generation throughout the supply chain.

### 7.6.1 iLUC impact

Table 7.8 presents the estimated contribution of GHG emissions from iLUC induced by bio-based feedstock supply to the potential Climate Change impact of partially bio-based and biodegradable mulching film.

**Table 7.8.** iLUC contribution to the potential Climate Change impact of mulching film LCA scenarios.

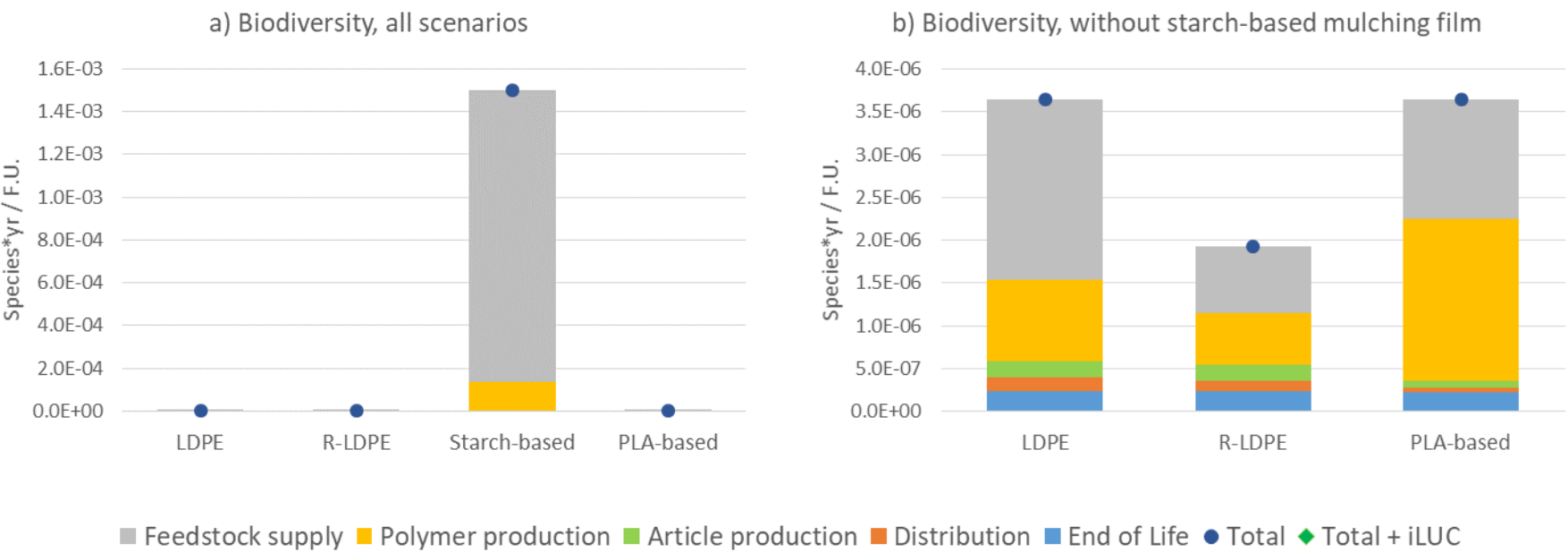
Scenario / Polymer	iLUC Climate Change impact [kg CO <sub>2</sub> eq/FU]	Total Climate Change impact (incl. iLUC) <sup>(1)</sup> [kg CO <sub>2</sub> eq/FU]
S1 – LDPE	-	(589)
S2 – R-LDPE	-	(399)
S3 – Starch/PBAT	2.35	542 (540)
S4 – PLA/PBAT	7.54	685 (678)

<sup>(1)</sup> Values in parenthesis refer to the Climate Change impact without the iLUC contribution.

### 7.6.2 Biodiversity impact

Estimated potential Biodiversity impacts of the investigated scenarios are presented in Figure 7.12. Impacts are separately presented by including and excluding starch-based mulching film, to allow appreciating differences among the other compared scenarios. It is reminded that the assessment is performed through an endpoint-level indicator relying on impact assessment methods that for some of the underlying midpoint impact drivers differ from those recommended in the present method.

1



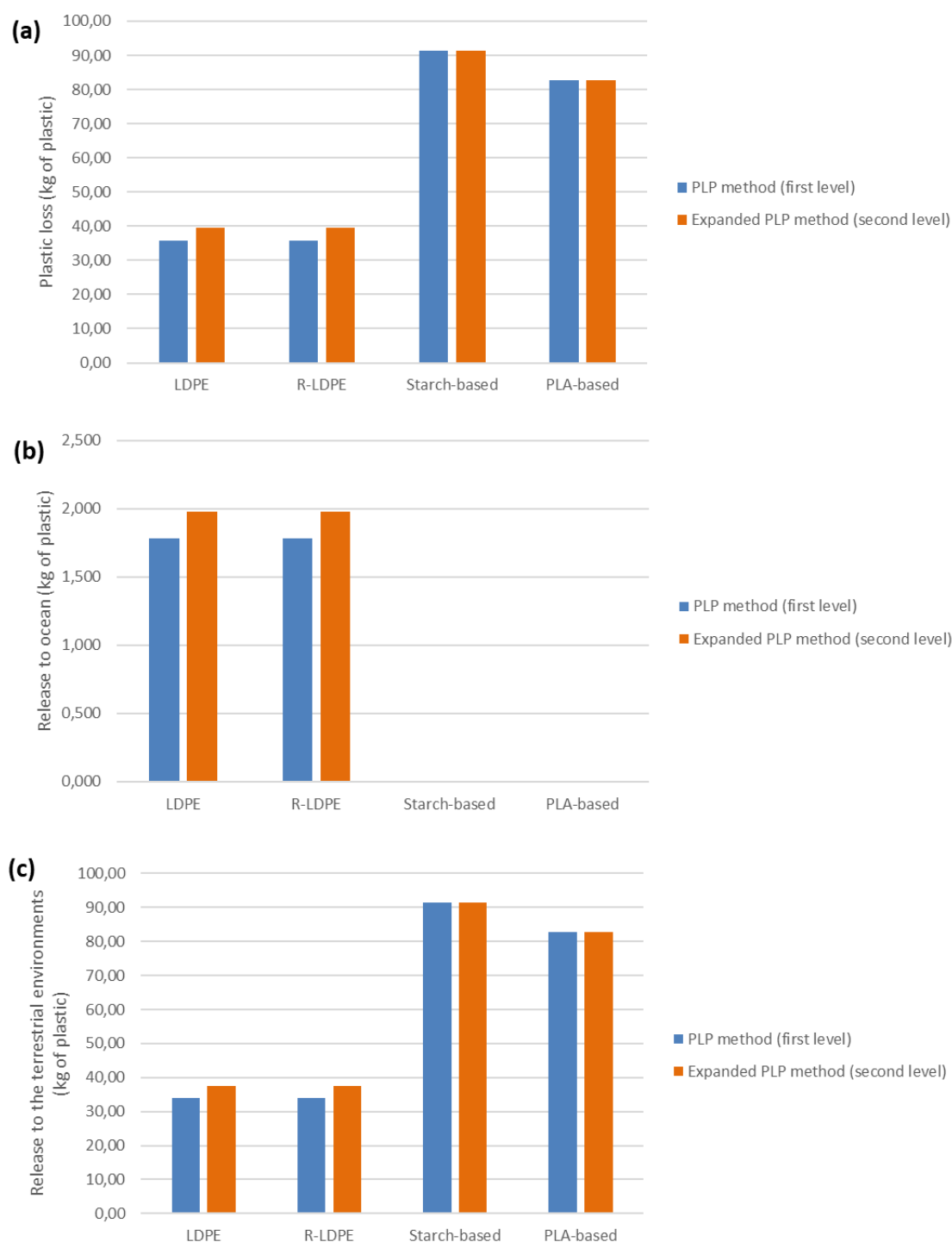
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3

**Figure 7.12.** Potential biodiversity impact of mulching film LCA scenarios, expressed as potential loss of species per year, including (a) and excluding (b) starch-based mulching film.

### **7.6.3 Macro- and micro-plastics generation**

Figure 7.13 shows the total plastic loss, release to ocean and release to the terrestrial environment estimated for the assessed mulching films scenarios, considering the whole product life cycle. Estimates obtained by applying all the two different explored approaches are presented, including the total loss and release to ocean/terrestrial environment calculated through both the *PLP method* (first level) and the *Expanded PLP method* (second level).

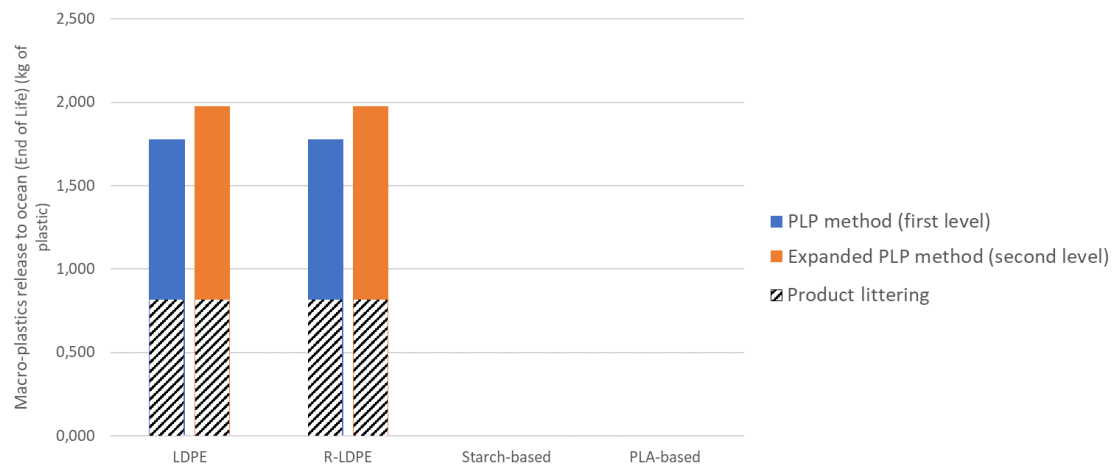
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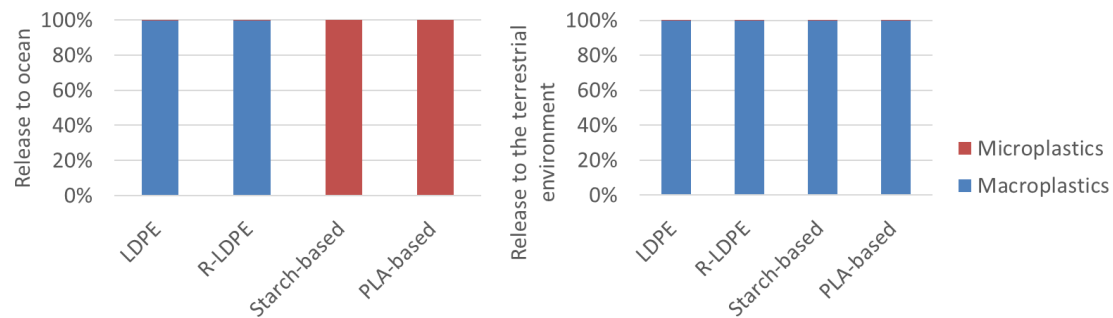
3 **Figure 7.13.** Total plastic loss (a), release to ocean (b) and release to the terrestrial environment  
4 (c) estimated for mulching films LCA scenarios with different approaches. The contribution of both  
5 macro- and micro-plastics is included, considering the whole product life cycle.

6 Regarding the sole release of macro-plastics to ocean at End of Life (from product  
7 littering and waste mismanagement), Figure 7.14 presents the contribution of each  
8 mulching films scenario, estimated with the two different explored approaches: the *PLP*  
9 *method* (first level), the *Expanded PLP method* (second level).



**Figure 7.14.** Macro-plastics release to ocean at End of Life, estimated with different approaches.

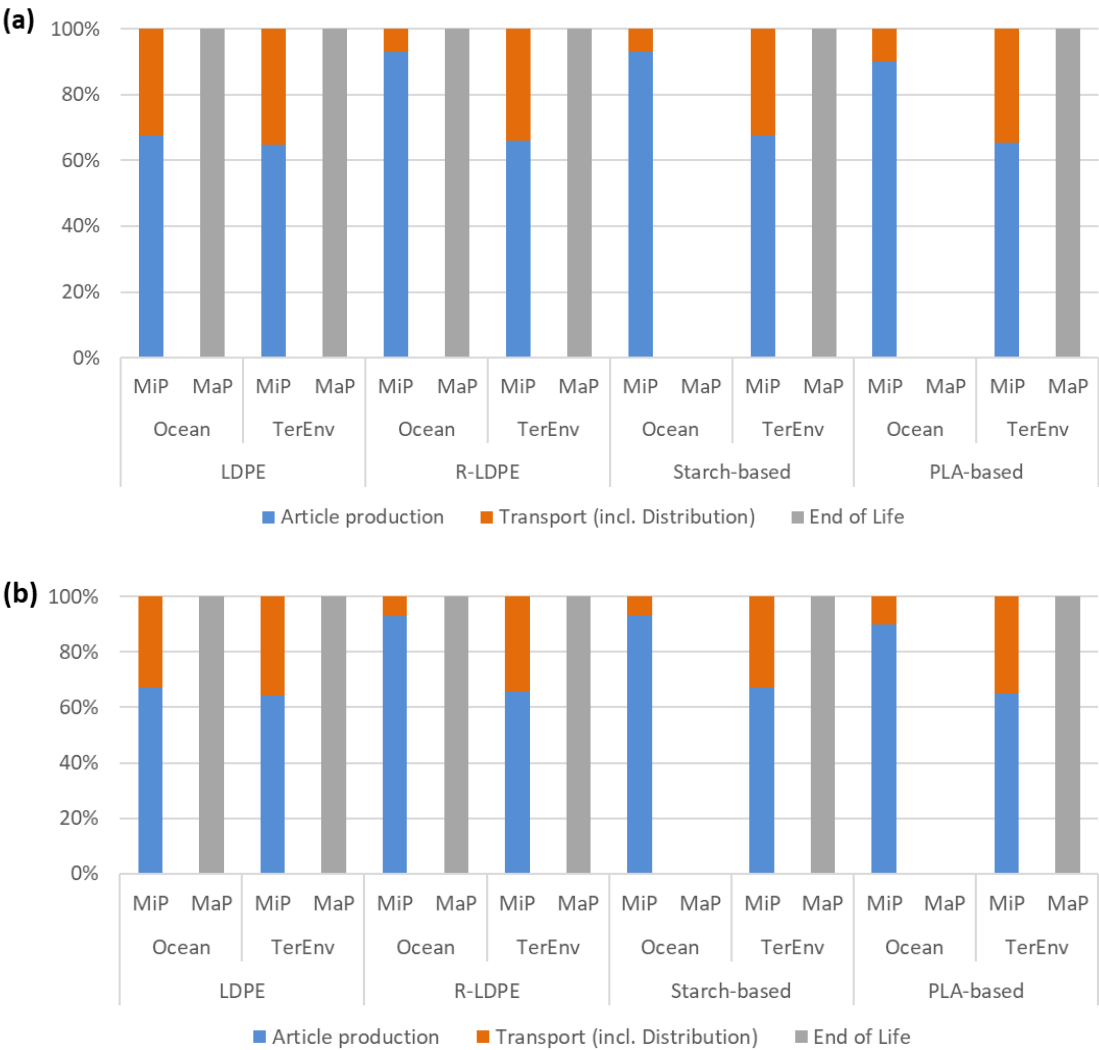
As for the share between the release of macro- and micro-plastics, Figure 7.15 shows as an example the relative contribution of the two categories to the total release to ocean and to the terrestrial environment estimated through the *PLP method* (first level). Note that the results are similar to those obtained when applying the *Expanded PLP method* (second level), as further discussed in Section 7.7.7.



**Figure 7.15.** Contribution of macro- and micro-plastics to the total plastic release to ocean and to the terrestrial environment estimated with the *PLP method* for each mulching films LCA scenario.

The contribution of the different lifecycle stages to the total macro- and micro-plastics release to ocean and to the terrestrial environment of the assessed mulching films scenarios has been finally evaluated. The results of this “hotspot analysis” are reported in Figure 7.16 for both the *PLP method* and the *Expanded PLP method*. Results are not shown for the other (sensitivity) approaches as they either cover only plastic losses or only a part of the product life cycle (i.e. End of Life).





**Figure 7.16.** Contribution of life cycle stages to the total macro-plastics (MaP) and micro-plastics (MiP) release to ocean (Ocean) and to the terrestrial environment (TerEnv) estimated for mulching films LCA scenarios with (a) the PLP method (first level) and (b) the Expanded PLP method (second level).

## 7.7 Interpretation

In the interpretation of case study results, most relevant impact categories and lifecycle stages of the studied systems are firstly identified (7.7.1 and 7.7.2, respectively). Scenario impacts are then compared, considering both characterised LCIA results (7.7.3) and total normalised and weighted impacts (7.7.4). Results obtained by applying individually each viable End of Life option are separately discussed in Section 7.7.6, while the effects of iLUC and the contribution to macro- and micro-plastics generation are addressed in Sections 7.7.5 and 7.7.7, respectively.

Most relevant processes were not identified, since it was not possible to achieve a similar level of vertical disaggregation for foreground processes included in the life cycle models of the analysed scenarios (e.g. a higher disaggregation could be achieved for Starch-based mulching films, but this was not the case of other scenarios). Therefore the identification of most relevant processes cannot be carried out consistently across all scenarios, and a more detailed investigation for specific scenarios would not be meaningful. The identification of most relevant elementary flows was also not undertaken, as this would require prior identification of most relevant processes. Note, however, that any company, organisation or any other supply chain actor applying the

present method shall proceed with the identification of both most relevant processes and elementary flows.

### 7.7.1 Identification of most relevant impact categories

Table 7.9 shows the most relevant impact categories identified for each mulching film scenario, based on normalised and weighted impacts, according to the approach outlined in the Method (Report I, Section 6.2.1). Relevant categories were identified as those that cumulatively contribute to at least 80% of the total normalised and weighted impact of the considered impact categories, excluding toxicity-related ones (these being still based on characterisation factors implemented in the EF 2.0 impact assessment methods, and hence not yet updated based on REACH data<sup>130</sup>). 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. Climate Change is the most relevant category for Recycled LDPE mulching film and for both bio-based alternatives, while for fossil-based LDPE film, Resource Use – fossils is the most relevant one. While these categories have a prominent role in all scenarios (covering 70-81% of the total impact), also other categories are identified as relevant. These include Particulate Matter (all scenarios except fossil-based LDPE film), and Acidification (limited to PLA-based film). For LDPE film, only Acidification is included. However, all of these categories show a limited contribution (lower than 6% each and not larger than 10% altogether).

**Table 7.9.** Most relevant impact categories identified for mulching film LCA scenarios and related contribution to the total normalised and weighted scenario impact.

S1 - Fossil-based LDPE		S2 – Recycled LDPE	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Resource Use - fossils	42.4%	Climate Change	49.4%
Climate Change	38.2%	Resource Use - fossils	28.5%
Particulate Matter	4.8%	Particulate Matter	6.0%
<b>Total</b>	<b>85.3%</b>	<b>Total</b>	<b>83.9%</b>
S3 – Starch-based		S5 – PLA-based	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Climate Change	44.5%	Climate Change	40.8%
Resource Use - fossils	33.0%	Resource Use - fossils	29.4%
Particulate Matter	4.5%	Particulate Matter	5.7%
		Acidification	5.5%
<b>Total</b>	<b>82.0%</b>	<b>Total</b>	<b>81.3%</b>

<sup>130</sup> According to the latest version of the Product Environmental Footprint Category Rules Guidance (EC, 2018), toxicity-related impact indicators based on these characterization factors shall be excluded from the procedure to identify most relevant impact categories.

## 7.7.2 Identification of most relevant life-cycle stages

Table 7.10 shows the most relevant lifecycle stages in the relevant impact categories identified in Section 7.7.1, and the associated contribution, quantified according to the rules detailed in the Method (Report I, Section 6.2.2). Most relevant stages include those that together contribute to at least 80% of the total characterised lifecycle impact in the specific category, and are highlighted in yellow in Table 7.10. For a more exhaustive picture, the contribution of all lifecycle stages is reported.

For fossil-based LDPE mulching film, Polymer Production and Feedstock Supply are identified as the most relevant life cycle stages both in terms of Resource Use – fossils and of Climate Change. Feedstock Supply covers almost 80% of the Resource Use – fossil impact, being the most relevant stage in this category. Conversely, in Climate Change Polymer Production is more relevant (46.4%), while the contribution of Feedstock Supply is more limited (22%) and End of Life is also relevant (18.7%).

Recycled LDPE film shows a similar picture to fossil-based LDPE for what concerns Resource Use – fossils and Climate Change, where the most relevant stages are Feedstock supply (83% of the total impact by itself) and Polymer Production (43%) respectively. However, additional relevant stages in terms of Climate Change impact are in this case represented by End of Life and Article Production, while Feedstock Supply is no longer relevant. As for Particulate Matter, the most relevant stage is End of Life, but its contribution is comparable to that of Polymer Production (nearly 30%) and the contribution of all the remaining life cycle stages (except Distribution) is needed to cover 80% of the total impact.

For both starch-based and PLA-based mulching film, Polymer Production represents the most relevant life cycle stage across all the identified most relevant categories and especially in Resource Use – fossils, where it is responsible for over 90% of the total impact. Its contribution is still prevailing also in the remaining categories, although in these cases also End of Life (Climate Change) or Feedstock Supply (Particulate Matter and Acidification) are identified as relevant stages. On the other hand, the stages of Article production and Distribution only provides a minor contribution across all the relevant categories, and are never identified as relevant.

**Table 7.10.** Contribution of life cycle stages to the total characterised impacts of the most relevant categories identified for mulching film LCA scenarios. Most relevant stages (i.e. those contributing to more than 80% of the total impact) are highlighted in yellow.

<b>S1 - Fossil-based LDPE</b>					
<b>Resource use, fossils</b>		<b>Climate Change</b>		<b>Particulate Matter</b>	
<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	78.9%	Polymer Production	46.4%	Polymer Production	31.7%
Polymer Production	29.7%	Feedstock Supply	21.9%	Feedstock Supply	24.1%
Article Production	5.9%	End of Life	18.7%	End of Life	19.0%
Distribution	2.7%	Article Production	8.4%	Article Production	12.7%
End of Life	-17.2%	Distribution	4.7%	Distribution	12.5%

1

S2 – Recycled LDPE					
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	42.6%	Feedstock Supply	83.2%	End of Life	29.0%
End of Life	27.5%	Polymer Production	42.5%	Polymer Production	27.5%
Article Production	12.5%	Article Production t	16.6%	Article Production	19.3%
Feedstock Supply	11.9%	Distribution	6.4%	Feedstock Supply	13.5%
Distribution	5.6%	End of Life	-48.9%	Distribution	10.6%
S3 – Starch-based					
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	74.0%	Polymer Production	92.2%	Polymer Production	67.4%
End of Life	17.0%	Article Production	4.7%	Feedstock Supply	20.2%
Article Production	4.5%	Distribution	1.7%	Article Production	8.0%
Feedstock Supply	2.5%	Feedstock Supply	1.4%	Distribution	4.4%
Distribution	2.0%	End of Life	0.0%	End of Life	0.0%
S4- PLA-based					
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	74.9%	Polymer Production	90.7%	Polymer Production	49.9%
End of Life	12.7%	Feedstock Supply	4.6%	Feedstock Supply	43.7%
Feedstock Supply	7.9%	Article Production	3.4%	Article Production	4.2%
Article Production	3.2%	Distribution	1.3%	Distribution	2.3%
Distribution	1.4%	End of Life	0.0%	End of Life	0.0%
Acidification					
<i>Life cycle stage</i>	<i>Contrib.</i>				
Polymer Production	48.5%				
Feedstock Supply	45.3%				
Article Production	3.1%				
Distribution	3.1%				
End of Life	0.0%				

2

### 3 7.7.3 Interpretation of characterised results

4 In this section, the characterised potential impacts of the assessed scenarios (Figures  
5 7.5-7.7 and Table B.4.5 in Annex B.4) are compared to evaluate the effects of using an  
6 alternative feedstock or material for a specific fossil-based polymer of reference used in  
7 mulching film manufacturing. Therefore, the impacts of 100% recycled LDPE, starch-

based and PLA-based mulching films are compared with those of virgin, fossil-based LDPE film considered as a reference.

The comparison focuses on the sixteen default impact categories considered in the Product Environmental Footprint (PEF) context and in this assessment. Potential impacts on Biodiversity are discussed separately, as being estimated via an endpoint indicator that is not recommended in the PEF framework, and partially relying on different impact assessment methods for the underlying midpoint-level impact categories. In the following comparison, differences between scenario impacts lower than 10% were not considered significant, in light of the uncertainty associated with the developed life cycle models and the applied impact assessment models. Moreover, any comparative considerations related to Human Toxicity and Ecotoxicity impact categories need to be interpreted considering the higher uncertainty of the underlying impact assessment models and results. The same applies, to a lower extent, also to Water Use and Land Use.

Focusing on the first alternative scenario (i.e. recycled LDPE mulching film), a lower impact is observed compared to virgin fossil-based LDPE film in the vast majority of the impact categories (i.e. all except Ozone Depletion). In particular, an impact reduction between 26% and 65% occur in the more stable categories (i.e. Climate change, Acidification, Eutrophication, Photochemical Ozone Formation, Particulate Matter, and Resource Use – fossil). Conversely, in Ozone Depletion the impact is increased by 34%, due to the higher impact associated with recycled feedstock supply (for reasons which could not be investigated further). However, results related to this impact category are less stable than those mentioned above, and need to be interpreted with more caution.

It is noted that the assumed recycled content for R-LDPE film is set to 100%, which does not yet appear to be feasible at present. Moreover, it has to be considered that the R-LDPE feedstock was not derived from closed loop recycling of agricultural mulching film, but from unspecified LDPE waste from municipal and industrial collection. This assumption disregards the effect of the contamination of used mulching film by soil (quantified in 45% of collected film by weight; see Section 7.4.5.2), which may play a key role in film recycling. Indeed the presence of soil may increase the impacts of the recycling process, due to additional energy consumption for handling and removal, and to the need of treating or disposing of the removed material.

Concerning the first bio-based and biodegradable alternative (i.e. starch-based mulching film), an improved environmental performance is observed in nearly half impact categories (7 over 16), with respect to virgin fossil-based LDPE film. These mainly include more stable categories such as Acidification, Eutrophication – terrestrial, Photochemical Ozone Formation, Particulate Matter and Resource Use – fossils, with an impact reduction between 13% and 39% (note that the three categories identified as most relevant are also included). This improvement mostly reflects the reduced mass of material required per functional unit (i.e. less than half compared to LDPE film), which compensate for the generally increased Polymer Production impacts. In most of the remaining categories, an impact increase is observed, especially in Land Use (+486%), and Eutrophication – freshwater (+300%), due to the substantially increased contribution of the Polymer Production stage and in many cases also of Feedstock Supply. However, a relevant increase is also observed for other less stable or more uncertain impact categories, including Ecotoxicity – freshwater (+480%), Human Toxicity – non-cancer (+160%), and Ozone Depletion (+194%). In Climate Change and Water Use, both alternatives are instead comparable. Compared to 100% recycled LDPE film, a worsened performance is observed for starch-based film, across the whole set of impact categories (showing an even relevant impact increase). However, the earlier considerations on the current feasibility of having a 100% recycled content in LDPE films holds true.

As for PLA-based biodegradable mulching film, an improved environmental performance compared to fossil-based LDPE film is observed only a limited number of impact categories, including Resource Use – fossils (identified as one of the most relevant; 25% impact reduction) and Photochemical Ozone formation (-16%), beyond the less stable Ozone Depletion (-36%), and Human Toxicity – cancer (-15%). In the remaining

categories, an impact increase is observed, despite the reduced mass of material required per functional unit compared to conventional non-biodegradable LDPE film. The only exception is Ionising radiation, where both alternatives are comparable.

Comparing the two partially bio-based and biodegradable mulching film alternatives, starch-based film exhibits a lower impact than its PLA-based counterpart in most of the categories (13 over 16), with the mass of both alternatives being comparable. Impact reductions achieved in the more stable categories mentioned above range between 19% and 58%. However, a worse performance is observed in terms of the less stable Ozone Depletion (+358%), and of the more uncertain Human Toxicity – non-cancer (+15%) and Resource Use – minerals and materials is (+16%). This is due to the higher impact of Polymer Production, which is responsible for the highest share of impacts of starch-based mulching film in almost all impact categories, and in many of these (more than half) production of additives used for Thermoplastic Starch provides an even important contribution, which can reach 50% of the total life cycle impact.

Focusing on estimated potential impacts on biodiversity, recycled LDPE mulching film shows a better performance than its virgin fossil-based counterpart (50% impact reduction), reflecting the lower burdens from Feedstock Supply and Polymer Production (which are less resource- and emission-intensive), rather than actual differences in supply-chain drivers towards biodiversity. Conversely, starch-based mulching film shows a disproportionately higher impact compared to all the other examined alternatives. This is due to the dramatically increased impact from (bio-based) Feedstock Supply, which mostly come from terrestrial ecotoxicity (endpoint) impacts from potato cultivation<sup>131</sup>, and, to a lower extent from wheat cultivation. However, the larger uncertainties of characterisation factors applied to the quantification of ecotoxicity impacts, and variability in the actual use of substances (pesticides) contributing to such impacts must be acknowledged. Moreover, it has to be reminded that direct potential biodiversity (ecosystem) impacts from oil leakage are not quantified for fossil-based mulching films considered as a reference (although emissions from leakage per unit of oil supplied are reported to be quite small; see Section 4.4.1.1). The impact of PLA-based mulching film is comparable with that of virgin fossil-based LDPE film, as the increased Polymer Production impact is balanced by a decreased contribution from Feedstock Supply, and the nearly halved mass of polymer required per functional unit also play a role. However, PLA-based film is outperformed by recycled LDPE film (showing a 90% higher impact than the latter).

#### **7.7.4 Interpretation of normalised and weighted results**

This section briefly compares the investigated scenarios based on the respective total impact scores, calculated by aggregating normalised and weighted impact assessment results across all impact categories (Table B.4.7 in Appendix B.4). The comparison is carried out according to the same criteria outlined in Section 7.7.3 (in terms of contrasted scenarios) and considering significant only impact differences larger than 10%. While affected by greater uncertainty (from the calculation and application of normalisation and weighting factors), total impact scores provides a more immediate and synthetic indication of the overall environmental performance of the analysed product scenarios. This facilitates the interpretation compared to looking at characterised results from several impact categories. However, such estimates are inevitably affected by value choices underlying the definition of weighting factors aimed at establishing an order of relevance for the different impact categories in a European decision context.

The comparison among total normalised and weighted scenario impacts reflects the main outcomes emerged when focusing on characterised impact assessment results in single impact categories, and can be summarised in the following points:

- Recycled LDPE mulching film shows a lower total impact than its virgin fossil-based counterpart (-50%) and the lowest among the assessed alternatives,

<sup>131</sup> Dominated by insecticide emissions to soil (which are responsible for more than 90% of the total impact).

thanks to the high (100%) recycled content assumed in this analysis and the resulting reduced impact from the stages of Feedstock Supply and Polymer Production.

- Starch-based biodegradable mulching film has a moderately reduced impact compared to virgin fossil-based LDPE film (-20%), reflecting the improved performance in all the impact categories identified as most relevant (Climate Change, Resource Use – fossils and Particulate Matter). However, 100% recycled LDPE film still provides a better overall performance (with starch-based film showing a 50% higher total impact).
- PLA-based biodegradable mulching film shows an overall impact comparable to that of virgin fossil-based LDPE film (9% increase), but higher than the other considered mulching film alternatives +40% compared to starch-based film, +110% compared to recycled LDPE film). This is mainly due to the increased impact of the Polymer Production and/or Feedstock Supply stages.

### 7.7.5 Effects of indirect land use change (iLUC)

When the contribution of indirect land use change is taken into account, only small changes are identified in the Climate Change impact of starch-based and PLA-based mulching film (i.e. +0.4% and +1.1% respectively). This is at least partly explained by the only partial bio-based content in the polymer (40-45%) and, in the case of PLA-based film, also by the relatively higher Climate Change impact associated with Polymer Production (and of the whole life cycle) even without considering the iLUC contribution.

Overall, accounting for iLUC effects does not affect the relative performance of the assessed scenarios (i.e. their comparison is not affected). However, this outcome needs to be red also in light of the GHG emission factors applied for iLUC in this assessment, which appear to fall in the lower end of the range of values available in the recent literature (see Section 4.4.16.3 of the Method (Report I)). Moreover, it should be kept in mind that the iLUC contribution considered in this case study only accounts for greenhouse gas emissions, meaning that only impacts on Climate Change are addressed. A larger iLUC GHG contribution and the inclusion of the additional use of converted land and/or intensification with the resulting nutrients-related emissions may thus worsen the Climate Change, Acidification, and Eutrophication impacts of the two bio-based mulching-film scenarios.

### 7.7.6 Alternative End of Life options

This section discusses characterised scenario results of non-biodegradable mulching film alternatives (i.e. virgin and recycled LDPE film) calculated by applying individually each viable End of Life option as an exclusive (100%) End of Life scenario (as presented in Figures 7.9 to 7.11). For biodegradable mulching films (i.e. starch-based and PLA,-based films) only in-situ biodegradation was considered to be viable, so that no specific considerations are reported below regarding these alternatives.

Note that the main purpose is to evaluate how the performances of single scenarios are affected by the change in the applied End of Life option, rather than to mutually compare the analysed mulching film scenarios for different End of Life options. However, the results should not be interpreted as a strict comparison among different End of Life alternatives for the studied article, since the evaluation applies a product perspective, and burdens/benefits of selected End of Life options (e.g. recycling) are partitioned between different product life cycles (which prevents from capturing the full implications of having a given waste stream routed to a specific End of Life option). In a waste management system perspective, where allocation is not needed, the total (system-wise) savings associated with the End of Life pathway “100% recycling” would likely be higher than what presented in this report where a product perspective is applied.

Due to the generally modest contribution of the End of Life stage to the overall scenario impacts, in several impact categories they are not significantly affected by the change in the applied End of Life option. However, within this possibly limited range of variation, the following considerations can be made.

None of the three considered product End of Life scenarios (i.e. mechanical recycling, incineration and landfilling) can be identified as preferable across all the assessed impact categories and mulching film scenarios. For both virgin and recycled LDPE mulching film, mechanical recycling is the preferable scenario in few impact categories, including Climate Change, Human Toxicity – cancer, Ecotoxicity – freshwater, Water use, and limited to virgin LDPE film, Human Toxicity – non-cancer and Resource Use – fossils. In the remaining categories, recycling is comparable with incineration, with both incineration and landfilling, or is in between these two options. In Ozone Depletion, the recycling scenario shows the worst performance.

Incineration of the two fossil-based mulching films (both virgin and recycled) has the worst performance only in terms of Climate Change and Water Use, while being the preferable scenario in a number of impact categories, including Ozone Depletion, Particulate Matter, Ionizing Radiation, Acidification, and Land Use. Limited to recycled LDPE film, incineration is also preferable in the case of Photochemical Ozone Formation, Eutrophication – terrestrial, Eutrophication – marine, Resource Use – minerals and metals, and Resource Use – fossils.

Landfilling is the worst scenario in the majority of the impact categories (13 over 16), in line with the priority order outlined in the “Waste Hierarchy”, which sets disposal as the least preferable option (EC, 2008). Exceptions are Climate Change, Ozone Depletion, and Water Use, where other End of Life scenarios show the highest impact, as previously discussed.

#### **7.7.7 Interpretation of results related to macro- and micro-plastics generation**

This section discusses the results presented in Section 7.6.3 on the estimated potential generation of macro- and micro-plastics of the assessed mulching films scenarios. The latter are compared according to the same criteria applied to the interpretation of the potential environmental impacts (Section 7.7.3), initially focusing on the results from the two-level approach applied as a base case. No sensitivity analyses were assessed for this case study.

Regarding the total loss of plastics (macro- and micro-plastics) and the resulting release to the terrestrial environment (Figure 7.13), the two base-case approaches (*PLP method* and *Expanded PLP method*) provide similar results when comparing the different mulching film scenarios. In both cases, biodegradable mulching films (starch-based and PLA-based) show a significantly higher plastic loss and release to the environment compared to the respective fossil-based counterparts (i.e. LDPE and R-LDPE film). Despite the nearly 50% lower amount of polymer required per functional unit by such biodegradable alternatives, the corresponding total loss and release is 250% (starch-based film) and 230% (PLA-based film) larger than the two non-biodegradable solutions considered as a reference. This is a consequence of the End of Life option applied to biodegradable mulching films, which after use are entirely left on the field for in-situ biodegradation (after being possibly incorporated into the soil during the next crop cycle). Therefore, 100% of the applied biodegradable plastic material was accounted as a loss and then release to the terrestrial environment (in contrast to non-biodegradable films, where only 10% of the material is left on the field, and hence released to soil, after removal). It must be noted, however, that the applied accounting framework excludes the effects of biodegradation of the product once released (or left) in the environment, due to lack of complete understanding of fragmentation and biodegradation pathways of biodegradable plastic products in the environment itself. The estimated release hence represents the immediate release to the environment of the product as such, without



reflecting the subsequent behaviour and fate of the material over time, which affect the ultimate release of any smaller plastic particles to the soil or other environmental compartments. On the other hand, scenarios relying on non-biodegradable, fossil-based mulching films show the highest release to ocean, as biodegradable mulching films left on the field and released to soil were not considered to reach the ocean.

No difference in the total plastic loss and release is observed between LDPE and recycled LDPE films. This is because the two scenarios require the same mass of material per functional unit, and the respective supply-chains imply similar transportation requirements. Such results highlight that changing the fossil-based feedstock does not affect the dominating contribution from macro-plastics loss and release at End of Life (see below for further discussion on this), while variations in micro-plastics generation due to different transport requirements along the life cycle (reflecting different feedstock or polymer origins) only marginally affect the overall results.

Comparing the two base-case approaches, the *Expanded PLP* method results in a 10% higher plastic loss and release to ocean and the terrestrial environment for the fossil-based scenarios compared to the original *PLP method* (Figure 7.13). This is mainly due to an increased contribution of macro-plastics loss at End of Life, which also accounts for mismanagement of plastic waste exported for (sub-standard) recycling outside the EU (where the overall waste mismanagement share is increased by 2% compared to the assumption that all plastic waste is recycled in Europe). In the case of bio-based scenarios, this increased on macro-plastics loss and release is not observed, as the products are not collected from the field and not treated through End of Life pathways. As a result, bio-based scenarios are not affected by plastic waste exported for recycling in non-EU countries. Conversely, the two additional micro-plastics sources considered in this alternative approach (road markings and marine coatings) does not significantly affect the results, due to the overall modest contribution of micro-plastics to the total release (in terms of mass).

Focusing on the relation between total loss and release (which depends on the parameters of the *PLP method*), the plastic loss is in this case modelled as fully released to the environment (i.e. loss and release are identical) due to the lack of informal waste collection for mulching film (which cannot be littered by the end consumer). In relative terms, release to ocean accounts for a marginal share of the plastic loss (5%), while release to the terrestrial environment is the most relevant (95%) for fossil-based mulching film scenarios. Regarding bio-based scenarios, 100% of the material left (i.e. "lost") on the field was assumed to be released to the terrestrial environment. As shown in Figure 7.15 for the *PLP method*, the total plastic release to both ocean and the terrestrial environment is dominated by the release of macro-plastics at the End of Life (more than 99%), while micro-plastics released throughout the upstream life cycle have only a minor role. Exceptionally, micro-plastics dominate the plastic release to ocean for bio-based scenarios. Similar results are also observed for the *Expanded PLP method* (not shown), despite two additional sources of micro-plastics are considered in the latter (i.e. road markings and marine coatings). In both cases, this is a consequence of the prevailing mass of macro-plastics, which is directly related to the mass of films required per functional unit (reference flow), by means of higher loss and release rates compared to micro-plastics. Therefore, the mass of macro-plastics is at least one order of magnitude higher than the mass of micro-plastics, which instead depends on the quantity of relevant lifecycle processes by means of (much) lower loss and release coefficients (depending on the source).

With regard to the origin of released plastics (in terms of life cycle stage and source), macro-plastics entirely originate from the End of Life stage (Figure 7.16), as they are only derived from mismanagement of the product as waste. Note that no littering is assumed for mulching films as it is considered that films are managed as part of cars at the End of Life thereby following specific collection and management pathways. On micro-plastics release, the relative contribution of the different sources (e.g. loss of plastic pellets and tire abrasion) depends on the characteristics of the supply-chain (e.g.

1 feedstock origin, which affect the contribution from transport across the life cycle), as  
2 well as on the mass of polymer required per functional unit (which affect, to different  
3 extents, all the considered sources). However, within the limited role of micro-plastics in  
4 the overall release to the environment, the most relevant contribution is generally  
5 provided by pellet losses from the Polymer and Article Production stages, while transport-  
6 related sources have a more restricted role.

7 In conclusion, the magnitude of plastic loss and subsequent immediate release to the  
8 environment associated with the mulching film life cycle mostly depends on their  
9 biodegradability properties and the consequently applied End of Life options. Due to the  
10 exclusion of the effects of biodegradation processes in the applied accounting framework  
11 (as discussed above), a best overall performance is associated to non-biodegradable,  
12 fossil-based alternatives in terms of plastic loss, total release and release to the  
13 terrestrial environment, since farmers collect the mulching from the field after  
14 cultivation. Conversely, non-biodegradable alternatives are associated to a larger plastic  
15 release to the ocean, since biodegradable ones only contribute to the release of micro-  
16 plastics to ocean (from upstream supply-chain processes), while macro-plastics are  
17 released exclusively to the terrestrial environment (i.e. the soil) when such alternatives  
18 are applied. From a methodological perspective, the different approaches provide similar  
19 results when comparing alternative scenarios.

## 8 Case study 5: Nursery pots

This case study focuses on plastic nursery pots where plants are grown in the nursery business of the agricultural sector. Nursery pots can serve a number of uses such as, starting seeds, growing young plants in optimal conditions (e.g. greenhouse) and transporting plants to a new location for transplanting. They are generally round or squared with holes at the bottom to allow excess water to flow out (Handreck et al., 2002) while, for seedlings starting in commercial greenhouses, pots usually take the form of trays with cells, each cell acting as a small pot. Pots are produced in a number of standard sizes, identified in the EU market, by a standard code referring to the main dimension (e.g. diameter at the top, in case of round pots).

The focus of the analysis is on standard round pots used at nurseries, so that squared pots and trays for seed starting are excluded. No further specification about colour or crop-specific function is provided. This choice allows to keep a broader scope, as well as to explore a wider range of materials and feedstock within a single case study.

### 8.1 Assessed scenarios

A number of scenarios were analysed to explore the potential impacts associated with the use of alternative types of feedstock or materials for the manufacturing of plastic pots (Table 8.1). Polymers traditionally used for pot production are PP and HDPE from (virgin) fossil-based feedstock sources, which were considered as reference materials for the comparison (Scenarios 1 and 2). The use of recycled PP and HDPE from separately collected, post-consumer plastic waste as a feedstock was then explored (Scenarios 3 and 4), by assuming 100% of the material input coming from recycling. Due to the absence of specific and representative data on the current average recycled content for pots at the EU level, the analysis focused on pots production entirely relying on recycled material. This allows to assess the effects of a complete substitution of the virgin material by its recycled counterpart, and to quantify the maximum potential environmental benefits or drawbacks expected from this replacement.

Bio-based, drop-in alternatives to fossil-based PP and HDPE were also assessed (Scenarios 5 and 6), considering ethanol derived from Brazilian sugarcane as a raw material for the production of both polymers (being sugarcane the most widely used feedstock at present). It is noted that another, fully bio-based polymer, i.e. Polylactic Acid (PLA), may also be used to produce (biodegradable) plastic pots, especially in combination with reinforcing fibres in composite materials (Schrader et al., 2016), which are beyond the scope of this projects. Moreover, these options are still at the research level and there is no prediction on when and if they will be available on the market. Therefore, they were not considered in this assessment.

Regarding End of Life, all currently viable treatment and disposal options were considered for each plastic pots material, including mechanical recycling, incineration and landfilling. As a base case, the impacts of each scenario were assessed with reference to an EU-average End of Life scenario including all the options currently applied at the EU-level. For all scenarios, these include all the viable options reported above, which were combined as described in Section 8.4.5.1 on End of Life modelling. In addition, scenario impacts were calculated by individually considering each currently viable End of Life option. Note that, despite bio-based PP and HDPE incorporate bio-based material, they are not biodegradable, as the final polymers have the same characteristics as their fossil-based counterpart ("drop-in" solution). Therefore, biological treatment options such as composting and anaerobic digestion are not viable for these materials.

1

**Table 8.1.** LCA scenarios assessed for nursery pots case study.

Scenario	Polymer	Monomer or Co-polymer	Feedstock	End of Life options <sup>(1)</sup>
1 - Conventional polymer 1	PP	Poly-Propylene	Fossil-based (crude oil/ natural gas)	Recycling Incineration Landfilling
2 - Conventional polymer 2	HDPE	Ethylene	Fossil-based (crude oil/ natural gas)	Recycling Incineration Landfilling
3 - Alternative polymer 1	R-PP (100% recycled content)	Poly-Propylene	Waste PP (post-consumer)	Recycling Incineration Landfilling
4 - Alternative polymer 2	R-HDPE (100% recycled content)	Ethylene	Waste HDPE (post-consumer)	Recycling Incineration Landfilling
5 - Alternative polymer 3	Bio-PP	Bio-Poly-Propylene	Sugarcane (BR)	Recycling Incineration Landfilling
6 - Alternative polymer 4	Bio-HDPE	Bio- Ethylene	Sugarcane (BR)	Recycling Incineration Landfilling

2  
3  
4

(1) The impacts of scenarios were individually assessed for each listed End of Life option, as well as for a combination of such options reflecting as far as possible the current (real or potential) average situation at the EU level.

## 5 8.2 Functional Unit and reference flow

6 Among the huge variety of nursery plastic pots available on the EU market, the analysis  
7 focuses on 10 cm standard round pots characterized by the following features:

- 8 - Upper diameter: 10 cm;
- 9 - Height: 9.5 cm;
- 10 - Lower diameter: 8.5 cm;
- 11 - 10 holes (diameter 1 cm) on the bottom for excess water leakage.

12 The main function of the studied article is to allow growing and moving of seedlings and  
13 young plants. The functional unit was thus defined as:

14 "growing and/or moving of one thousand young plants by means of thousand 10 cm  
15 standard round pots, in the EU, without breaking, and ensuring the plant to properly  
16 develop until the transplanting in a new location" (Table 8.2).

17

1 **Table 8.2.** Definition of the functional unit for pots LCA scenarios.

Aspect	Description
“What” (function(s) or service(s) provided)	Growing and moving of young plants
“How much” (extent of the function(s) or service(s))	1000 young plants (one per pot)
“How well” (expected level of quality)	Without breaking during the transport, and ensuring proper development of the plant
“How long” (duration/lifetime of the function or service)	One time
“Where” (location/geography of the service)	In the EU

2 The reference flow pertaining to each scenario (i.e. the amount of polymer required to  
3 fulfil the functional unit) was based on the mass of pots of the relevant material. This  
4 was calculated considering pot dimensions (as specified above), an average thickness for  
5 single use plastic pots (i.e. 0.1 cm), and the corresponding material density (Table 8.3).

6 **Table 8.3.** Calculation of the reference flow for nursery pots LCA scenarios.

Polymer	Polymer volume <sup>(1)</sup> (cm <sup>3</sup> /pot)	Density (g/cm <sup>3</sup> )	Pot mass (g/pot)	Reference flow (kg/FU)
PP (all types of feedstock)	18.01	0.908 (0.89-0.92) <sup>(2)</sup>	16.35	16.35
HDPE (all types of feedstock)	18.01	0.950 (0.93-0.97) <sup>(2)</sup>	17.11	17.11

7 <sup>(1)</sup> Calculated on the basis of pots dimensions and average thickness (i.e. 0.1 cm)

8 <sup>(2)</sup> <https://www.plasticseurope.org/en/about-plastics/what-are-plastics/large-family/polyolefins>

### 9 **8.3 System boundary**

10 In all scenarios, the system boundary was set in order to cover the most relevant stages  
11 and processes of the full product life cycle on a cradle-to-grave perspective, as described  
12 below and depicted in Figures 8.1 to 8.6:

- 13 • *Feedstock Supply*<sup>132</sup> – covering extraction, transport and possible refining of crude  
14 oil and natural gas (fossil-based polymers), collection, transport and sorting of  
15 plastic waste (recycled polymers), crop cultivation (bio-based polymers), as well  
16 as transport of these feedstock sources to downstream conversion processes (e.g.  
17 naphtha cracking, polymer recycling, sugarcane fermentation);
- 18 • *Polymer Production*<sup>133</sup> – covering all the activities associated with the conversion  
19 of feedstock materials into the relevant monomer(s) and final polymer, including  
20 any transport among these activities and final transport of polymer granulate to  
21 downstream manufacturing processes;

<sup>132</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

<sup>133</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

- 1 • *Article Production*<sup>134</sup> – including pots manufacturing through injection moulding of  
2 polymer granulates;
- 3 • *Distribution* – including transport of pots from the manufacturing site to retailers  
4 and from these to final consumers;
- 5 • *End of Life* – covering collection, transport, recycling, incineration, or disposal of  
6 pots after use, including any avoided processes from virgin material or energy  
7 substitution.

8 A different nomenclature was applied for some life cycle stages compared to the default  
9 nomenclature specified in the Method (and in the PEF framework) to make it more  
10 relevant for the investigated supply chains and the project scope. Moreover, the default  
11 “Raw Material Acquisition and Pre-processing” stage was further split into two separate  
12 sub-stages (i.e. Feedstock Supply and Polymer Production), to allow disaggregating the  
13 impacts of feedstock supply from downstream conversion processes, and hence to better  
14 appreciate any differences among the use of different feedstock sources (in line with the  
15 project goal).

16 The Use stage was excluded in this LCA study, as it can be reasonably assumed that the  
17 different pots materials investigated in this study does not affect activities related to the  
18 Use stage. Therefore, these activities can be excluded from a comparative assessment,  
19 as being identical for all the examined product systems.

20 Finally, it has to be noted that additives were not included in the assessment, due to lack  
21 of complete and consistent data on the use of additives in the production of nursery pots,  
22 of the examined polymers and for plastics in general, as well as on their release and fate  
23 over the product life cycle. This is acknowledged as a limitation of the study, as additive  
24 production can account for a non-negligible portion of cradle-to-gate Climate Change  
25 impact and energy demand of polymers, which is up to 46% for (starch-based) polymer  
26 grades including larger shares of additives (Broeren et al., 2017). Moreover, additives  
27 can also be relevant at the End of Life stage, where they can be released, as such or  
28 after degradation/conversion into different compound(s), in the environment (e.g. the  
29 soil in case of biodegradable plastics routed to biological treatments or subject to in-situ  
30 degradation).

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<sup>134</sup> Corresponding to the default stage of “Manufacturing” specified in the Method and in the PEF framework.

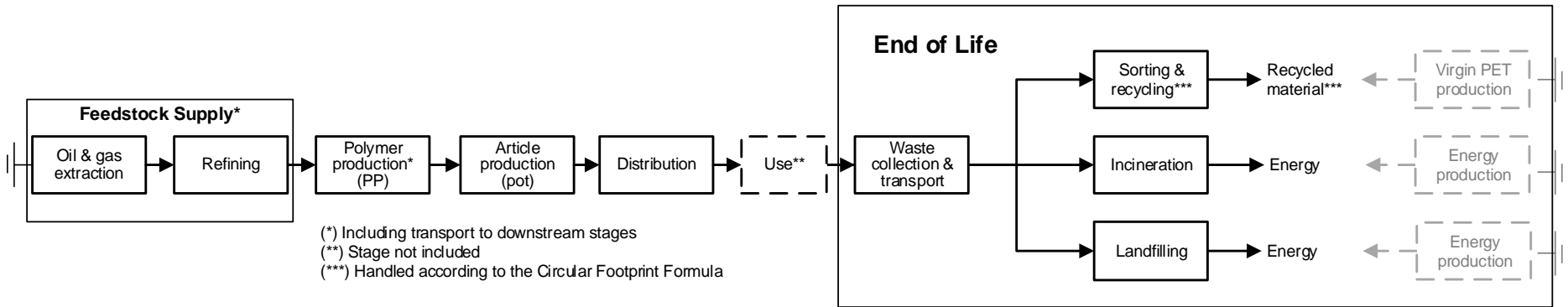


Figure 8.1. System boundary for fossil-based PP pots (Scenario 1).

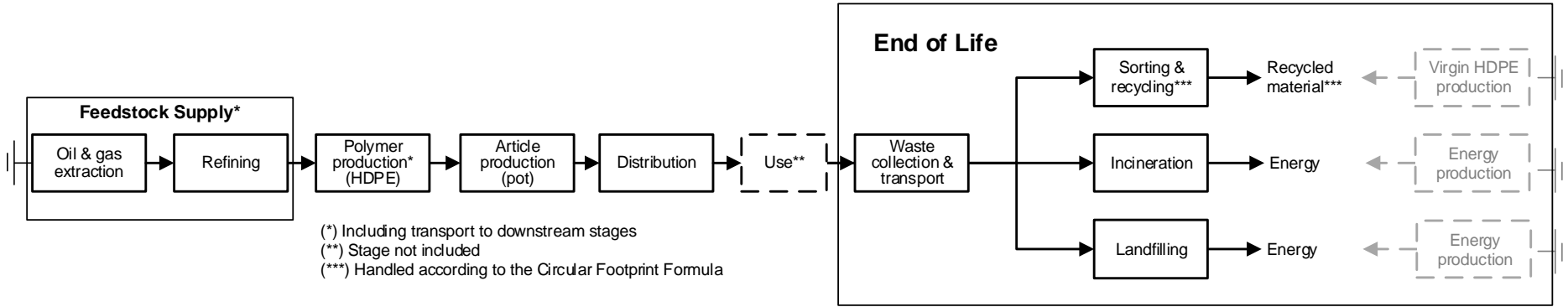


Figure 8.2. System boundary for fossil-based HDPE pots (Scenario 2).

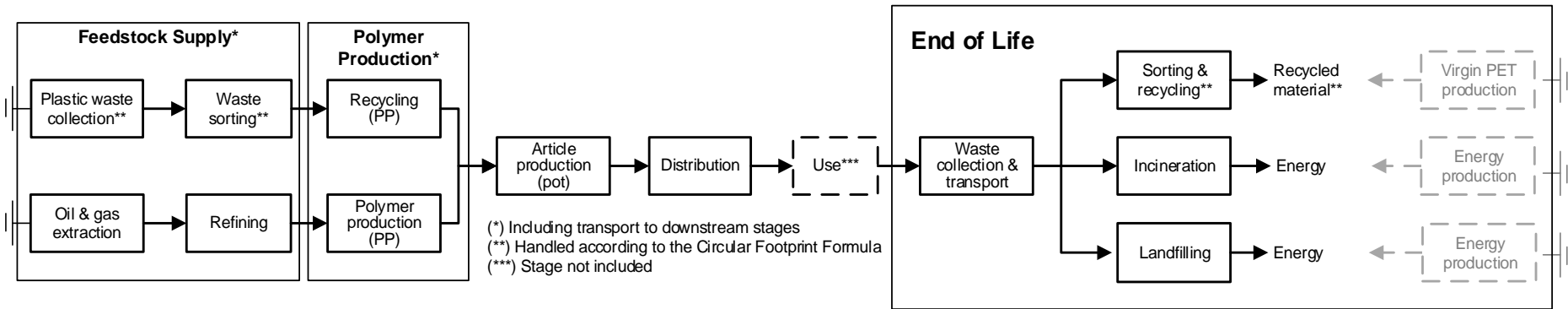


Figure 8.3. System boundary for recycled PP pots (Scenario 3).

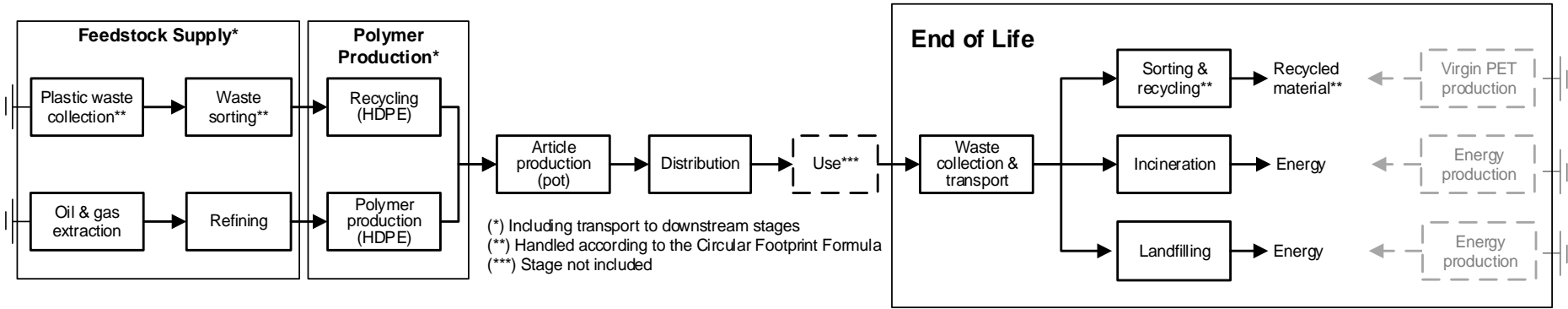


Figure 8.4. System boundary for recycled HDPE pots (Scenario 4).



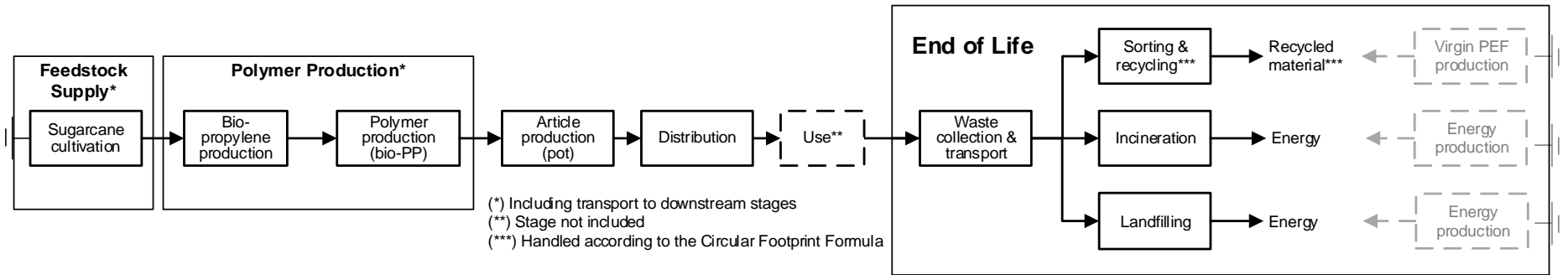


Figure 8.5. System boundary for bio-based PP pots (Scenario 5).

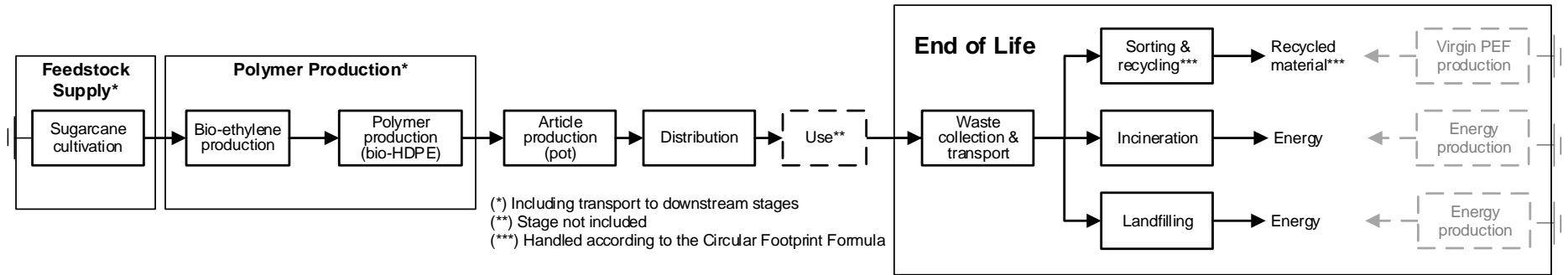


Figure 8.6. System boundary for bio-based HDPE pots (Scenario 6).

## **8.4 Life Cycle Inventory**

This section describes the overall approach for developing the Life Cycle Inventory of the analysed scenarios, along with the related assumptions and data sources. The description is separated by major lifecycle stages, in the following sub-sections (8.4.1 – 8.4.6). The list of processes, related data sources, and main modelling details are provided in Tables B.5.1 to B.5.6 in Annex B.5.

### **8.4.1 Feedstock Supply Stage**

#### **8.4.1.1 Fossil-based polymers**

For fossil-based polymers (HDPE and PP), the stage of Feedstock Supply includes the processes of crude-oil and natural gas extraction and transport, naphtha production in crude oil refineries, and its subsequent transport to downstream conversion processes (i.e. naphtha cracking). The modelling of these activities was carried out as described in Section 4.4.1.1, where the reader is referred to for further details on data sources, covered sub-processes and related environmental burdens, main modelling choices (e.g. allocation) and any adjustments performed in the implementation of the applied datasets.

#### **8.4.1.2 Recycled polymers**

For recycled polymers (R-HDPE and R-PP), Feedstock Supply consists of collection of post-consumer plastic waste of the relevant polymer, and its subsequent transport and sorting in specific facilities. These processes were modelled as already described in Section 4.4.5 for the Beverage Bottle case study, and refer to collection and transport of separately collected plastic waste at the municipal level (in the absence of specific data for plastic waste collection from the different sectors it may come from), as well as to sorting of mixed plastic waste in dedicated facilities. Both processes were implemented in the lifecycle model according to the Circular Footprint Formula (CFF), which is the default approach to handle recycling situations in the Product Environmental Footprint (PEF) context. According to this approach, only 50% of the burdens from collection, transport, sorting and recycling were assigned to the recycled content in pots ( $A=0.5$  for HDPE and PP used in unspecified applications), the rest being assigned to the system providing material for recycling. Further details on the implementation of the CFF are provided in Section 4.4.2.2 on recycled polymers production.

#### **8.4.1.3 Bio-based polymers**

For bio-based polymers (Bio-HDPE and Bio-PP), the stage of Feedstock Supply includes cultivation of sugarcane in Brazil, and its subsequent transport to further processing in the same country.

Sugarcane cultivation was modelled through aggregated datasets from the GaBi database. Used in combination, these datasets depict a situation where 45% of sugarcane is manually harvested via the "slash and burn" practice, i.e. sugarcane residues (tops and leaves) are burned on standing plants before harvesting. This share was considered representative of the current situation. However, the slash and burn 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). Transport of harvested sugarcane to further processing in sugarcane mills for Ethanol 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).

## **8.4.2 Polymer Production Stage**

The Polymer Production stage covers the activities of feedstock processing into relevant intermediates and monomer(s), the polymerisation process, as well as any transport among these activities and final transport of polymer granulate to the pots manufacturing site. The following subsections (8.4.2.1 – 8.4.2.4) describe how these activities have been modelled in the present case study.

### **8.4.2.1 Fossil-based polymers**

For conventional, fossil-based polymers (HDPE and PP), the whole process chain from feedstock processing to polymerisation, through the production of intermediates and monomers, was modelled by means of aggregated, gate-to-gate datasets provided by Thinkstep. Inputs include combinations of crude oil, natural gas and naphtha, depending on the polymer. All conversion processes are assumed to take place in Europe, so that the datasets reflect EU average background conditions in terms of e.g. energy generation, material supply and transport. For both polymers, the main conversion process involved in the supply chain is steam cracking of naphtha and natural gas, delivering the monomers Ethylene and Propylene, along with Butadiene and other relevant intermediate flows (e.g. pyrolysis gas including mixtures of Benzene, Toluene and Xylenes). The inventory of this process is mainly based on industry data, completed, where necessary, by literature data. 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 co-products. The final polymerisation process of HDPE and PP is mainly modelled based on industry data from internationally adopted production processes, integrated with literature data when needed. No allocation is performed for this stage, being HDPE and PP the only outputs from the respective production process.

### **8.4.2.2 Recycled polymers**

The production of recycled HDPE granulates out of sorted, post-consumer HDPE waste was modelled building upon the *ecoinvent* dataset “[*Europe without Switzerland*] *Polyethylene production, high density, granulates, recycled*” (no specific EF-compliant dataset is available for modelling purposes). However, the latter was updated according to the values and (where needed) flows, reported in a most recent and expanded version of the original data source (i.e. Franklin Associates, 2018, updating Franklin Associates, 2011) and determined as mass-weighted averages of data collected from several recycling facilities in the United States. Updated exchanges were then combined with EF background datasets for energy and material supply under EU-average conditions. The overall recycling efficiency is equal to 84%, with removed contaminants and process waste being sent to incineration (which along with co-combustion in cement kilns is one of the two typical fate of plastic recycling residues with high calorific value; Rigamonti et al., 2014).

Data from the abovementioned source (i.e. Franklin Associates, 2018) were also used to develop a new inventory dataset related to the production of recycled PP granulate from sorted, post-consumer PP waste. Similarly to recycled HDPE resin, such data represent mass-weighted averages of values collected from several recycling plants in the United States, and for modelling purposes were used in combination with EF background datasets for energy and material supply under EU-average conditions. In this case, the overall recycling efficiency is equal to 85.5%, with removed contaminants and process waste still sent to incineration (consistently with the assumption in the HDPE recycling process described above, and the typical fate of plastic recycling residues).

According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the recycling process were allocated to the recycled content in pots ( $A = 0.5$  for HDPE and PP used in unspecified applications). However, the recycled content carries 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 life cycle providing the recycled material). Assuming a value of the  $Q_s/Q_p$  factor equal to 0.9 (similarly to the value recommended for HDPE and PP when used in packaging applications, in the absence of more specific data) the allocated share of virgin production impacts is equal to 45% ( $A \times Q_s/Q_p = 0.5 \times 0.9 = 0.45$ ). Virgin polymer production burdens were modelled as described above (Section 8.4.2.1) for conventional, fossil-based HDPE and PP, and in Section 8.4.1.1 for the respective Feedstock Supply.

### 8.4.2.3 Bio-based polymers

#### 8.4.2.3.1 Bio-based HDPE

For bio-based HDPE production, no datasets are available from commercial databases or the set of EF-compliant datasets. All conversion processes 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 considers Brazil as a relevant geography. 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 bioethanol-based polymers investigated in this project (i.e. Bio-PET, Bio-LDPE and Bio-PP), and to better align to the hierarchy for the handling of multi-functionality adopted in the present 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 metals<sup>135</sup>. The specific amount of sugarcane required for Ethanol production is equal to 15 kg per kg of Ethanol.

Ethanol produced in Brazil was assumed to be transported to Europe for further conversion and polymerisation, consistently with the assumption performed in the aggregated, gate-to-gate datasets used to model the production of the other bioethanol-based polymers considered in this project. Transport is modelled according to the default transport scenario specified in the PEF context (and in the 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 Ethanol 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.com<sup>136</sup>, 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 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 producers of bio-based HDPE is located in Brazil. In this specific case, the whole process chain of conversion and polymerisation would take place in Brazil, with the final HDPE resin being eventually transported to

<sup>135</sup> Infrastructure processes related to the Ethanol 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 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.

<sup>136</sup> Available at: <https://www.searates.com/services/distances-time/>

Europe. The assumption performed 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), and hence a higher mass of material is transported per functional unit compared to the transport of the final HDPE resin.

The inventory for 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 report were combined with background EF datasets for energy generation, and *ecoinvent* background datasets for material production<sup>137</sup>. Reported transport activities were not implemented in the inventory, being transport of Ethanol (the main raw material) accounted for separately in the foreground inventory, while transport of the other input materials is included in the datasets applied to model their supply. Beyond Ethylene, a small amount of naphtha is also obtained as a co-product, which is handled via direct substitution of naphtha from crude oil refinery (modelled as described for the “Feedstock Supply” stage in Section 4.4.1.1). The data source does not report the specific Ethanol requirement, which was thus determined as the average of the values reported in IEA-ETSAP & IRENA (2013; 1.74 kg Ethanol/kg Ethylene) and IfBB (2018; 2.08 kg Ethanol/kg Ethylene), corresponding to an estimated consumption of 1.91 kg Ethanol per kg of Ethylene.

The final polymerisation step of Ethylene to HDPE was modelled based on data from the most recent PlasticsEurope ecoprofile (PlasticsEurope, 2016), as implemented in the *ecoinvent* database. Indeed, inventory data limited to the polymerisation stage 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 this process across all the investigated HDPE-based scenarios (i.e. fossil, recycled, and bio-based HDPE pots). The applied data represent average values of data collected from several European production units operated by PlasticsEurope member companies, covering 68% of 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 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 production<sup>138</sup>. No allocation or 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 (2017), i.e. 1.07 kg Ethylene/kg, representative of suspension polymerisation only.

#### 8.4.2.3.2 Bio-based PP

The production of bio-based PP out of Brazilian sugarcane was modelled based on an aggregated dataset provided by Thinkstep. The dataset covers the steps of sugarcane processing to bioethanol in Brazil (fermentation and distillation), bioethanol transport to EU via transoceanic ship, conversion to bio-ethylene (via dehydration) and subsequently to propylene (via dimerization and metathesis), as well as final polymerisation to PP. Combustion of bagasse from sugarcane processing generates surplus energy (electricity and heat) which were assumed to directly replace average electricity from the Brazilian

<sup>137</sup> 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.

<sup>138</sup> 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 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] *Market for chemical, organic*), i.e. Acetic Acid, Methanol, Urea, Vinyl Acetate, Ethylene Dichloride, and Formaldehyde). The respective shares were then equally subdivided among remaining chemicals.

grid, and thermal energy produced in the same country from natural gas. For the remaining conversion and polymerisation processes, no allocation or substitution needs to be applied. The inventory is mainly based on industry data from internationally adopted production processes, completed, where necessary, by literature data. The final (gas phase) polymerisation step specifically relies on secondary data. Background inventories from the GaBi database are used to model the burdens of the inventoried inputs and outputs.

#### **8.4.2.4 Transport of polymer granulate to the article production site**

Modelling of transport of polymer granulate from the polymerisation plant (inside or outside the EU), to the pots manufacturing site in Europe, is based on the default transport scenario (distances and vehicle types) specified in the PEF context (and in the 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 HDPE and PP), the following routes were thus considered:

- (i) 130 km by lorry (total weight >32 t; Euro 4);
- (ii) 240 km by train (average freight); and
- (iii) 270 km by ship (barge).

For the imported share of fossil-based HDPE and PP (22% and 10%, respectively), a transoceanic ship transport was considered as the main transport route. The corresponding overall 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)<sup>139</sup>. Countries contributing to at least 90% of the overall import were considered in the calculation, for an overall distance equal to 7,982 km for fossil-based HDPE, and to 8,609 km for fossil-based PP (see Tables 8.4 and 8.5). Oceanic ship transport was complemented with road transport to and from the harbour in both the exporting and importing country (i.e. the EU). Road transport is made by lorry (total weight >32 t; Euro 4) along an overall default distance of 1000 km.

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

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<sup>139</sup> Available at: <https://www.searates.com/services/distances-time/>

**Table 8.4.** Calculation of the overall average sea distance for imports of fossil-based HDPE to Europe.

Exporting country	Import <sup>(1)</sup> (%)	Import (% cum.)	Distance <sup>(2)</sup> (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>

<sup>(1)</sup> Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

<sup>(2)</sup> 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 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.

1 **Table 8.5.** Calculation of the overall average sea distance for imports of fossil-based PP to Europe.

Exporting country	Import <sup>(1)</sup> (%)	Import (% cum.)	Distance <sup>(2)</sup> (km)	Weighted distance (km)
SAUDI ARABIA	41.3	41.3	8767.28	3622
KOREA, REPUBLIC OF (SOUTH KOREA)	13.5	54.8	16702.31	2259
INDIA	7.15	62.0	10267.28	734
RUSSIAN FEDERATION (RUSSIA)	6.75	68.7	5574.44	376
BRAZIL	6.63	75.4	10107.97	670
ISRAEL (GAZA and JERICOHO- >1994)	4.86	80.2	2987.48	145
SOUTH AFRICA (incl. NA - >1989)	4.26	84.5	12160.92	518
EGYPT	3.20	87.7	3212.14	103
UNITED STATES	2.99	90.7	6061.9	181
Other countries	9.32	100.0	-	-
Overall weighted distance				8,609

2 <sup>(1)</sup> Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a).  
3 Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext  
4 data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries  
5 during a given year has been recalculated as the sum of imports from individual countries (to remove any  
6 discrepancies with the rounded total reported in raw Comext data).

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

### 12 8.4.3 Article Production Stage

13 Both HDPE and PP pots (from all types of investigated feedstock) were assumed to be  
14 manufactured through injection moulding, which is the most commonly applied process  
15 to convert raw polymer granules into gardening and nursery pots. These must not be  
16 confused with seedling nursery trays, which are typically manufactured via  
17 thermoforming. In the injection moulding process, melted or plasticized plastic material  
18 is injected by force into a clamped mould cavity (consisting of a single cavity or a number  
19 of similar or dissimilar cavities), where it is finally shaped into the desired object (Rosato  
20 et al., 2004).

21 The burdens of the overall conversion process were modelled through the aggregated, EF  
22 dataset [EU-28+EFTA] Injection moulding; plastic injection moulding | production mix, at  
23 plant | for PP, HDPE and PE {ec9ca75e-abdb-4d2e-9e18-ca1f5709a76d}", which  
24 accounts for a 97.1% conversion efficiency. Process losses (e.g. flawed pots) were  
25 assumed to be entirely recycled in external facilities via re-granulation into new polymer  
26 pellets, ultimately replacing virgin granules of the same material. Therefore, recycled  
27 HDPE granules were assumed to replace virgin, fossil-based HDPE granules, being the  
28 estimated share of bio-based HDPE on the market currently very low (i.e. 0.2%).  
29 Similarly, recycled PP granules were assumed to replace virgin, fossil-based PP granules,  
30 since no bio-based PP was reported to be available on the market at the time of  
31 developing this analysis (EUBP, 2019). The recycling process and the resulting virgin  
32 material substitution were modelled based on the same data as End of Life recycling of  
33 sorted, post-consumer HDPE and PP pots, in the absence of more specific data for  
34 recycling of pre-consumer, industrial scraps. For further detail on the modelling of



recycling and avoided virgin material production (including the implementation of the Circular Footprint Formula), the reader is hence referred to Section 8.4.5 on End of Life modelling.

#### **8.4.4 Distribution Stage**

The transport of pots from the manufacturing site to the final user was modelled based on the default transport scenario specified in the PEF context (and in the Method) for the route *factory → retail → final client*. The following routes were thus considered:

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

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

#### **8.4.5 End of Life Stage**

This section describes the modelling of the End of Life stage (EoL) of nursery pots. In particular, Section 8.4.5.1 focuses on the definition of the EU-average End of Life scenario, which is considered as a base case for the calculation of the potential impacts of the compared LCA scenarios. The remaining sections (8.4.5.2 – 8.4.5.5) address the modelling of waste collection and transport, and of the different End of Life options explored in the study. Finally, Section 8.4.5.6 provides case-specific details on the estimate of the potential contribution of nursery pots to macro-plastic formation at the End of Life (including product litter) and micro-plastic generation throughout the supply chain.

##### **8.4.5.1 End of Life scenario**

Viable End of Life options for post-consumer pots made out of both HDPE and PP include mechanical recycling, incineration and landfilling. The combination of these alternatives into the average End of Life scenario considered for modelling purposes was defined according to available statistics for overall plastic waste management in the EU (EC, 2018), in the absence of more specific and suitable data for pots. This scenario was applied regardless of the feedstock used for polymer production (which do not affect viable End of Life options) and include 30% mechanical recycling (after separate collection and sorting), 39% incineration, and 31% landfilling.

##### **8.4.5.2 Modelling of waste collection and transport**

After use by citizens, waste pots are (separately) collected and transported to the different considered End of Life options. These activities were modelled according to the pathways, vehicle types and distances reported in Rigamonti et al. (2013) for plastic waste separately collected for recycling, and residual waste sent to incineration or landfilling. These data refer to a region with a well-developed municipal waste management scheme in northern Italy, but can be considered representative of several regions in Europe, where good levels of separate collection are achieved with the implementation of kerbside collection systems. Moreover, while the considered separate collection pathways and data may be more representative of packaging waste collection, they can be reasonably extended also to pots discarded by municipal users. Note, however, that since the modelling of collection and transport was implemented consistently across all the investigated scenarios, their comparison is not affected by the possible approximations and assumptions reported above (but only their “absolute” impact is). More information on the main assumptions and modelling details of the considered collection pathways is available in Section 4.4.5.2. Note that when EF

incineration and landfilling datasets were used, no transport of collected waste was separately modelled, since the respective burdens are already accounted for in such datasets. In this case, only collection was modelled, according to the approach described above.

#### **8.4.5.3 Modelling of sorting and recycling**

Before recycling, bales of separately collected plastic waste containing post-consumer pots are firstly 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.

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 data collected from different dual-stream and single-stream sorting facilities in the Country). For implementation in the model, the reported input/output data were complemented with background EF datasets representative of EU-average conditions, as better detailed in Section 4.4.5.3 (Table 4.9). A 100% sorting efficiency was assumed for waste pots, which entirely consists of one of the targeted materials for recycling. Moreover, no burdens from the treatment of any impurities sorted out as rejects were assigned to the same pots, to avoid falsely “punishing” them with burdens from unrelated product waste (and mostly coming from incorrect citizen behaviour). Therefore, the modelled sorting process only accounted for the material and energy inputs required to carry out this waste treatment activity.

For mechanical recycling of both HDPE and PP plastic products, no specific datasets are available in the PEF context. Therefore, for HDPE-based pot recycling, the *ecoinvent* dataset “[Europe without Switzerland] Polyethylene production, high density, granulate, recycled” was used as a basis for modelling purposes, in combination with EF background datasets for energy and material supply under EU-average conditions. Since a most recent and expanded version of the original inventory data source is available (i.e. Franklin Associates, 2018, updating Franklin Associates, 2011), the dataset was adjusted according to the updated exchange values and, if needed, exchange types, reported in the mentioned 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 being sent to incineration along with process waste. This is in line with the typical fate of recycling residues, which due to their high calorific value are normally sent to incineration or co-combustion in cement kilns (Rigamonti et al., 2014).

Similarly to HDPE pots, a new dataset was developed for mechanical recycling of PP-based pots, based on foreground inventory data available as well in Franklin Associates (2018)<sup>140</sup>. These still represent mass-weighted average values of point data collected from several US recycling facilities, which for modelling purposed were here combined with EF background datasets for energy and material supply (under EU-average conditions). The inventory accounts for an overall recycling efficiency equal to 85.5%, with removed contaminants and process waste being sent to incineration, consistently with the typical fate of plastic recycling residues, and the assumption performed in the HDPE recycling process described above.

Recycled polymer granulate was assumed to replace virgin granulate of the same material, whose primary production burdens were credited to the system. For polymers

<sup>140</sup> An aggregated EF-compliant dataset is also available for PP recycling in the US (based on an older version of the same inventory data source, i.e. Franklin Associates, 2011). However, since this could not be adjusted to reflect EU background conditions (due to its aggregated nature) and most recent data are available, a new disaggregated dataset was developed.

having both a fossil-based and a bio-based alternative available on the market as of today (i.e. only HDPE in this case study) 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 only 0.2% for HDPE (which is for 99.8% fossil-based HDPE)<sup>141</sup>. To account for the lower overall quality of recycled polymers compared to the replaced virgin polymers, a substitution ratio equal to 0.9 was considered for HDPE and PP, according to default values specified in the PEF context for such materials when used in packaging applications (in the absence of more specific data). To model the burdens of avoided primary production, the same datasets (or combination of datasets) used for the modelling of upstream production of the relevant polymers were applied (as described in Sections 4.4.2, and 4.4.1 for the related feedstock). This was made for consistency reasons, and to avoid possible distortions by applying different datasets from other sources.

According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the sorting and recycling processes were allocated to the system ( $A = 0.5$  for both PE and PP used in unspecified applications). Similarly, only 50% of the benefits from avoided virgin material production were assigned to the system itself.

#### **8.4.5.4 Modelling of incineration**

For conventional fossil-based polymers (i.e. HDPE and PP) aggregated, material-specific incineration datasets are available from the EF database, and were applied to model the treatment of pots made of these materials in a municipal waste incineration plant. Similarly, for bio-based HDPE and bio-based PP pots, partially aggregated, material-specific inventories from the GaBi database were applied (no EF datasets are available for these polymers).

All the selected datasets are developed based on a waste-specific incineration model, which has been described more in detail in Section 4.4.5.4. 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 general approach to handle energy recovery situations adopted in the PEF context and in the Method (Report I), the product system generating the waste material sent to incineration (i.e. the pots life cycle, in this case) takes the full burdens from the incineration process. However, it is credited for 100% of the benefits from avoided production of conventional energy (electricity and heat) replaced by energy recovered from waste. While in EF incineration datasets these credits are already accounted for in the aggregated inventory, for GaBi datasets they were added to the main process inventory. Therefore, the EU residual electricity grid mix (as modelled in the EF 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 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 includes 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. Also, thermal energy from waste (11%) was excluded, as according to the approach described above for the handling of energy

<sup>141</sup> The share of bio-based HDPE was estimated based on the global production capacity of bio-based PE in 2018 (European Bioplastic, 2019; 200 kt) and of PE as a whole in 2016 (PlasticsInsight, 2019; 103 Mt).

recovery situations, 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).

#### **8.4.5.5 Modelling of landfilling**

Landfilling of pots made of conventional non-biodegradable polymers (i.e. fossil-based HDPE and PP, both virgin and recycled) was modelled based on a common, aggregated EF dataset representing disposal of plastic waste in a managed, municipal solid waste landfill ([EU-28+EFTA] *Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific sites)*). The underlying inventory is material-specific, but refers to the average chemical composition and degradability values of generic plastic waste, rather than to those of the specific polymer. 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<sup>142</sup>. 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 emissions to the environment over a 100 year time horizon. Emissions occurring beyond 100 years are not accounted for in the model. Landfill gas generation is calculated based on the organic carbon content in the waste and the respective degradation rate assumed over 100 years (not reported), while, 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 rate of landfill gas, energy efficiencies of engines, collection rate of leachate and respective treatment efficiencies). Further details on how these parameters were set to reflect the EU-average situation are provided in Section 4.4.5.5.

The same dataset described above for landfilling of generic plastic waste was also employed as a proxy for landfilling of pots made of “drop-in”, non-biodegradable, bio-based polymers, i.e. bio-based HDPE and bio-based PP. Compared to the original dataset, emissions of CO<sub>2</sub> and CH<sub>4</sub> 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 degradation and those coming from background activities, the entire amounts of fossil CO<sub>2</sub> and CH<sub>4</sub> reported as released to air were converted to biogenic emissions. This approximation is considered acceptable, as reported CO<sub>2</sub> and CH<sub>4</sub> emissions are modest, and overall amounting to only 1.4% of the carbon content in the landfilled plastic material, despite they also include the contribution of emissions from background processes. This can be partly explained by the generally low degradation (and mineralisation) of conventional, non-biodegradable polymers in landfill, which is typically assumed to be in the range of 1% over 100 years (Doka, 2009b). On the other hand, it is acknowledged that this approach is in favour of the two bio-based polymers (i.e. Bio-HDPE and Bio-PP).

In line with the time horizon applied for landfill emission modelling in the selected dataset, (biogenic) carbon in polymers that is not degraded (mineralised) after 100 years from deposition was considered to be never released from the landfill body. As a base case, this missed release of atmospheric carbon taken up during biomass growth is not reflected in Climate Change LCIA results for bio-based HDPE and PP pots, since characterisation factors for biogenic CO<sub>2</sub> flows have been set to zero (fully conforming to prescriptions in the PEF method). However, to show the relevance of this choice on the

<sup>142</sup> While more polymer-specific landfilling datasets are available from other databases (i.e. *ecoinvent*), the mentioned EF dataset for landfilling of generic plastic was selected, as specifically referring to EU as the reference geography (in contrast to available polymer-specific datasets), and to keep consistency with the overall dataset selection “hierarchy” followed in this project (Section 3.5).

overall results, the Climate Change impact of the two bio-based alternatives was also calculated by acknowledging the effects of non-released biogenic carbon. For this purpose, a specific CO<sub>2</sub> uptake was modelled in the inventory, expressing the net amount of CO<sub>2</sub> taken up during biomass growth and not released (as CO<sub>2</sub> or CH<sub>4</sub>) during polymer degradation in landfill. The uptake was calculated based on the biogenic carbon content in the polymer, and assuming a mineralisation rate over 100 years equal to 1% for both polymers (Doka, 2009b). While the mineralisation rate assumed in the dataset is not reported explicitly, it seems to be in line with this assumption (according to the figures reported above regarding the inventoried carbon emissions). Considering a biogenic carbon content equal to 81.9% for both Bio-HDPE and Bio-PP, a carbon uptake equal to 2.97 kg CO<sub>2</sub>/kg bio-based polymer was calculated. This uptake was then characterised with a factor of -1 kg CO<sub>2</sub> eq. per kg CO<sub>2</sub>-C not degraded.

#### **8.4.5.6 Contribution to macro- and micro-plastics generation (including product litter)**

The contribution to macro- and micro-plastics generation of the analysed nursery pots scenarios was estimated according to the framework outlined in Annex B of the Method (Report I), and summarised in Section 4.4.5.6 of this report in terms of underlying quantification methods and approaches. In this section, the case-specific parameters considered to apply the latter to the assessed nursery pot LCA scenarios are reported.

##### **8.4.5.6.1 PLP method**

Regarding the estimate of the loss and release of macro-plastics at the End of Life stage (from product littering and waste mismanagement), the product-specific parameters reported in Table 8.6 were considered to apply the first level of the framework ("*PLP method*") to nursery pots (derived from Peano et al., 2020). Note that for the scenarios representing HDPE release rates are lower due to a higher economic value of the material.

**Table 8.6.** Case-specific parameters of the PLP method applied to the nursery pots LCA scenarios.

Parameter <sup>(1)</sup>	Value (HDPE, R-HDPE, Bio-HDPE)	Value (PP, R-PP, Bio-PP, PLA)
Littering rate (LR <sub>lit</sub> ) (%)	0	0
Release rate to ocean (Rel <sub>ocean</sub> ) (%)	1	5
Release rate to the terrestrial environment (Rel <sub>terenv</sub> ) (%)	1	95

<sup>(1)</sup> For details on the meaning of each parameter, refer to Annex B of the Method (Report I).

No case-specific parameters are required to calculate the supply-chain loss and release of micro-plastics to ocean and to the terrestrial environment by applying the *PLP method*. Indeed, the contribution of the considered micro-plastics sources (pellet loss and tire abrasion) depends on the magnitude of the relevant life cycle processes (i.e. polymer production and road transport) in the specific nursery pots scenario (which in turns depend on the reference flow<sup>143</sup> of each scenario). However, the quantification is made by means of default parameters that are not affected by the type of product, polymer or feedstock source (as reported in Annex B of the Method (Report I)).

##### **8.4.5.6.2 Expanded PLP method**

The parameters required to calculate the micro-plastics loss and release from marine coatings and road markings are not case-specific, and are reported in Annex B of the Method (Report I). Similar to the other considered micro-plastics sources, the

<sup>143</sup> The reference flow is the mass of pot material required to fulfil the functional unit.

contribution of these additional sources depends on the intensity of relevant (transport) processes in the specific product inventory, by means of the above-mentioned case-unspecific parameters.

As for the calculation of the additional contribution of plastic waste exported for recycling to the loss and release of macro-plastics from waste mismanagement, the case-unspecific parameters described in Annex B of the Method (Report I) were applied, in combination with the EU-average recycling rates estimated for each type of nursery pots (Section 8.4.5.1). For the sake of clarity, the applied recycling rates are summarised again in Table 8.7. Note that the bio-based scenario considering the use of PLA assumes a share of 15% of the pots to be composted instead of recycled.

**Table 8.7.** EU-average recycling rates considered for each nursery pots scenario to estimate the amount of waste plastic pots exported to non-EU countries for recycling.

Polymer	Recycling rate
HDPE	30%
R-HDPE	
Bio-HDPE	
PP	
R-PP	
Bio-PP	
PLA	15%

#### 8.4.5.6.3 Sensitivity analysis 1: alternative calculation of the total plastic loss along the supply-chain

The contribution of each (foreground) life cycle process to the total plastic loss of the different nursery pots scenarios was quantified by applying the approach by Ciroth & Kouame (2019) to the corresponding *ecoinvent* dataset or, if needed, adapted dataset from the same database<sup>144</sup>. Table 8.8 summarises the datasets considered for the cradle-to-gate process-chain involved in the production of each polymer, along with the possible adaptations performed. The datasets considered for the remaining foreground or background lifecycle processes are instead listed in Annex D Note that this approach excludes the contribution from product littering and waste mismanagement at End of Life.

<sup>144</sup> Equivalent to those applied in the actual lifecycle model of each scenario, which could not be directly considered for calculation purposes as the mentioned approach has been specifically developed to be applied to *ecoinvent* inventories.

**Table 8.8.** Ecoinvent datasets applied and possible adaptations performed to quantify the total plastic loss from the production of the different polymers considered for nursery pot manufacturing, through the approach developed by Citroth and Kouame (2019).

Polymer	Dataset	Adaptation
HDPE	[RER] Polyethylene production, high density, granulate	No adaptation required.
R-HDPE	Generic recycled plastic dataset developed on purpose.	No adaptation required.
Bio-HDPE	[RER] Polyethylene production, high density, granulate	90% of the original fossil feedstock is replaced with sugarcane.
PP	[RER] Polypropylene production, high density, granulate	No adaptation required.
R-PP	Generic recycled plastic dataset developed on purpose.	No adaptation required.
Bio-PP	[RER] Polypropylene production, high density, granulate	30% of original propene is substituted by bio-based propane from various residual bio-oils
PLA	[RER] Polylactide production, granulate, pot grade	No adaptation required.

#### 8.4.5.6.4 Sensitivity analysis 2: alternative calculation of macro-plastics release to ocean

This sensitivity analysis is not applied to this case study due to the lack of data for the consumption of nursery pots or broader product categories including the assessed product, associated to the limited granularity of the PRODCOM database (Eurostat, 2019c).

### 8.4.6 Calculation of iLUC impacts

As a base case, the contribution of Indirect Land Use Change (iLUC) to the potential Climate Change impact of bio-based pot alternatives was calculated according to the approach outlined in Section 4.4.16.4 of the Method (Report I).

In order to apply (recalculated) iLUC GHG emission factors from the EU 2015/1513 Directive (EC, 2015), the specific land demand for the production of relevant crops ( $m^2 \cdot year / kg$  crop) was calculated first. The calculation was based on the aggregated amount of arable and agriculture land occupation flows reported, for the relevant country (e.g. Brazil for sugarcane), in the dataset used to model the production of the specific crop. If the geography of such flows was not specified, all the arable and agricultural land occupation flows reported in the dataset were aggregated. These estimates were checked against values of land demand calculated based on 5-years average crop yields from FAOSTAT (FAO, 2019), which were found to be generally aligned with the former (absolute variation in the order of 15% for sugarcane used as a feedstock for both Bio-HDPE and Bio-PP). Hence, the values estimated based on land occupation flows were applied, for consistency with the overall LCI modelling of scenarios.

Specific land demand for crop production was then converted into a demand per functional unit, based on the specific consumption of crop for polymer production ( $kg$  crop/ $kg$  polymer, consistently with the values applied in the LCI modelling) and the amount of polymer needed to fulfil the functional unit (reference flow) in the relevant scenario. The iLUC contribution to the Climate Change impact was finally calculated by applying to the latter the recalculated GHG emission factors from the Directive. The described calculation steps to estimate the iLUC contribution to the potential Climate Change impact are summarised in Table 8.9.

1 **Table 8.9.** Calculation of the iLUC contribution to the potential Climate Change impact of nursery pots LCA scenarios relying on bio-based polymers.

Scenario / Polymer	Feedstock	Land demand for crop production <sup>(1)</sup> [m <sup>2</sup> *y/kg <sub>crop</sub> ]	Crop demand for polymer production [kg <sub>crop</sub> /kg <sub>polymer</sub> ]	Polymer demand per F.U. [kg <sub>polymer</sub> /FU]	iLUC GHG emission factor [kg CO <sub>2</sub> eq./ (m <sup>2</sup> *y)]	iLUC climate change impact [kg CO <sub>2</sub> eq./FU]
S5 – Bio-HDPE	Sugarcane (BR)	0.117 (0.135)	28.6	17.62	0.176	10.38
S6 – Bio-PP	Sugarcane (BR)	0.117 (0.135)	25.7	16.84	0.176	8.90

(1) Based on arable and agriculture land occupation exchanges reported in the dataset applied to the modelling of the specific crop. Values in parenthesis refer to land demand calculated based on crop yield data from FAOSTAT (5-years average), and are reported as a reference.



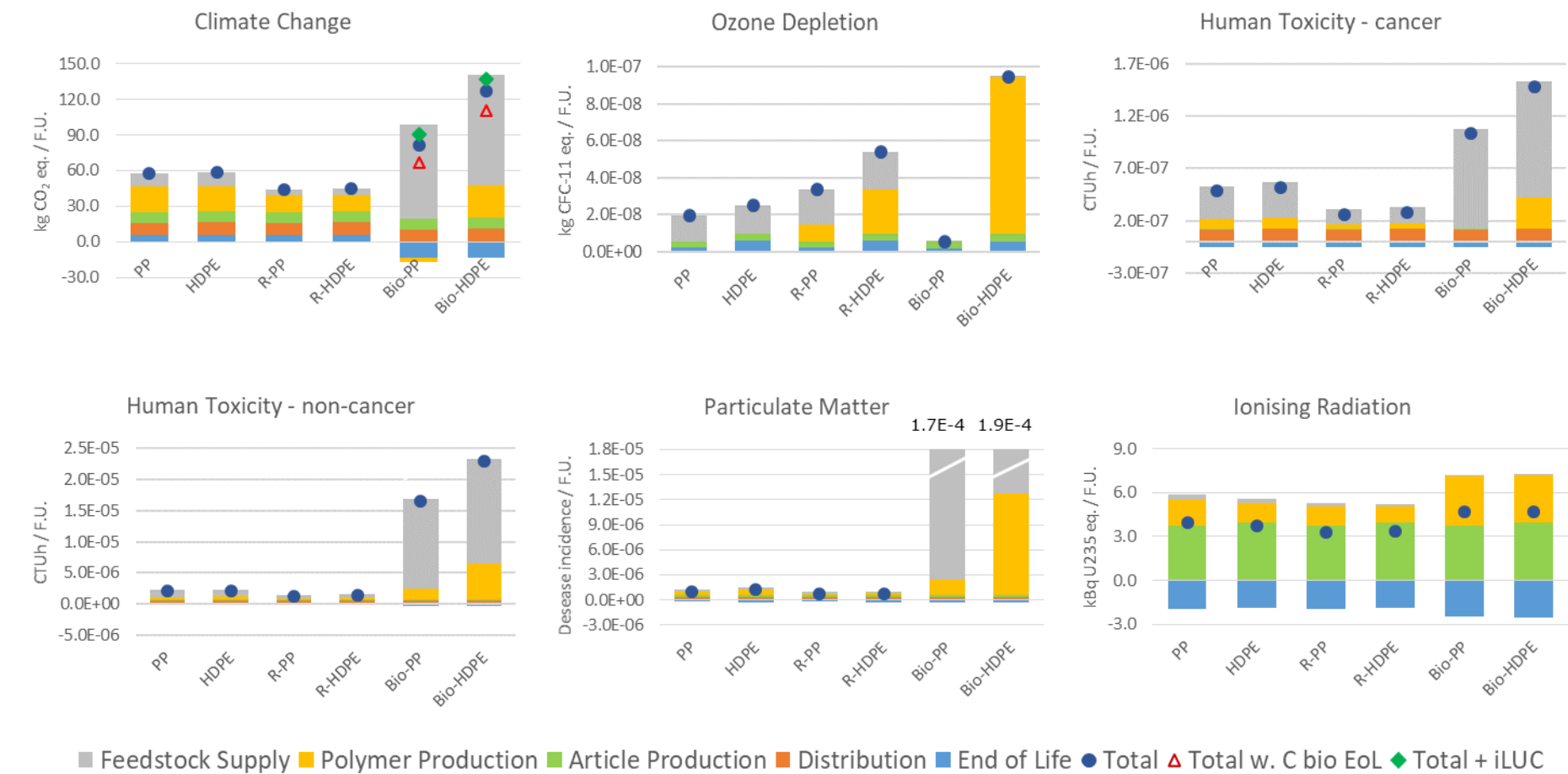
## 8.5 Life Cycle Impact Assessment results

The characterised potential impacts of the examined scenarios for nursery pots (with and without the iLUC contribution to Climate Change) are reported in Figures 8.7 to 8.9. These also show the breakdown of contributions from the main lifecycle stages, which include:

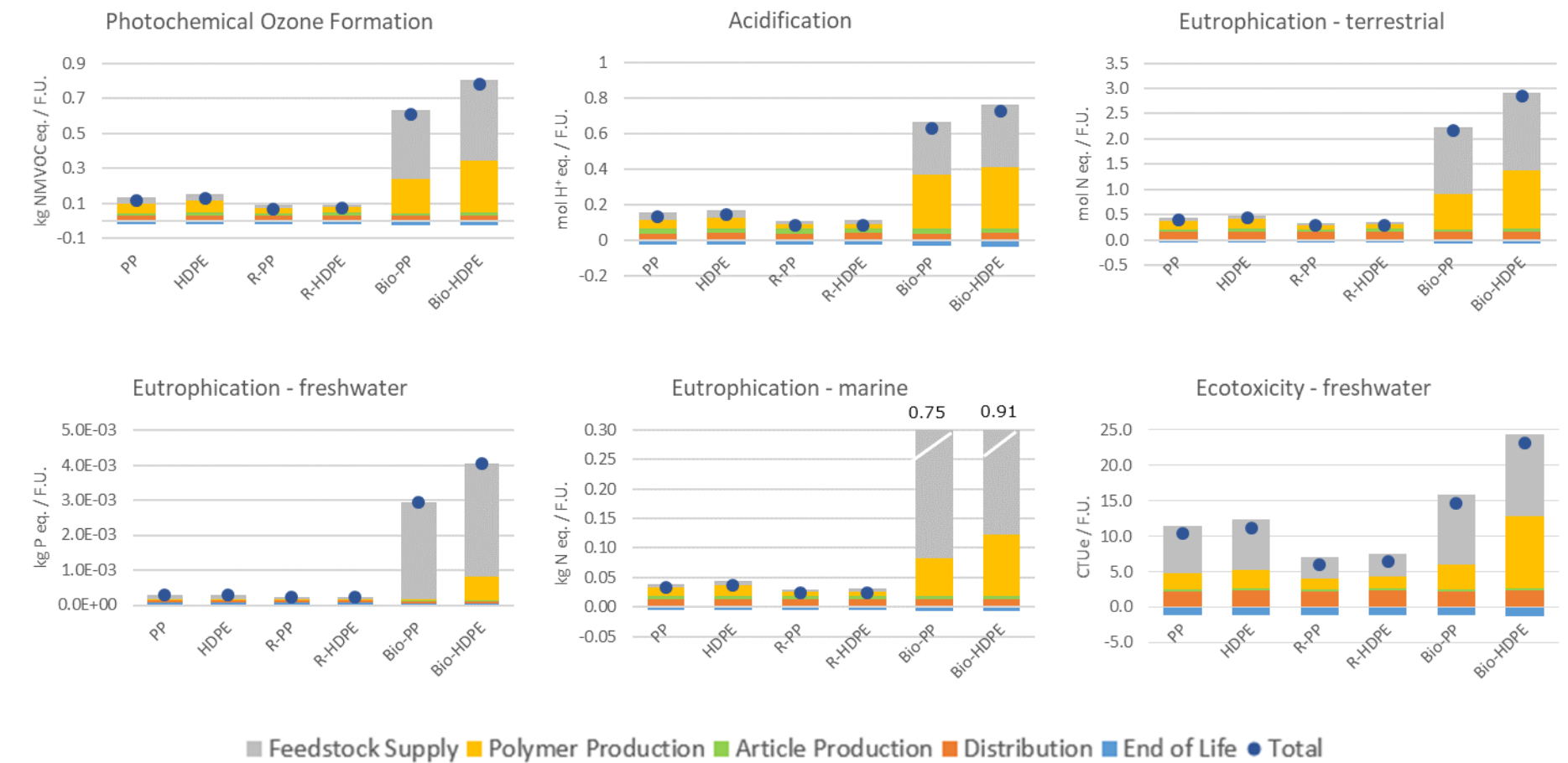
- i) *Feedstock Supply*, i.e. oil/natural gas extraction, transport and possible refining, transport of naphtha from refinery to downstream user; collection, transport and sorting of post-consumer plastic waste (recycled polymers); or crop cultivation and transport to further processing (bio-based polymers);
- ii) *Polymer Production*, i.e. all gate-to-gate activities from feedstock processing to monomer/intermediate production and polymerisation, including any transport among these and transport of polymer granulate to the nursery pots manufacturing site;
- iii) *Article Production*, i.e. conversion of the polymer into nursery pots by injection moulding;
- iv) *Distribution*, i.e. transport of nursery pots from the manufacturing site to the final user; and
- v) *End of Life*, i.e. waste pots collection, transport and treatment or disposal, as well as any avoided processes from downstream displacement of virgin materials and energy.

Figure 8.10 shows the contribution of each sub-category to the Climate Change impact (i.e. from fossil, biogenic and land use/land transformation GHG emissions.), while the overall characterised, normalised and weighted impact assessment results are available in Annex B.5. The latter are not reported in this section to keep readability of the document, despite this is not in alignment with the position suggested in the LCA report template provided in the Method (Report I).

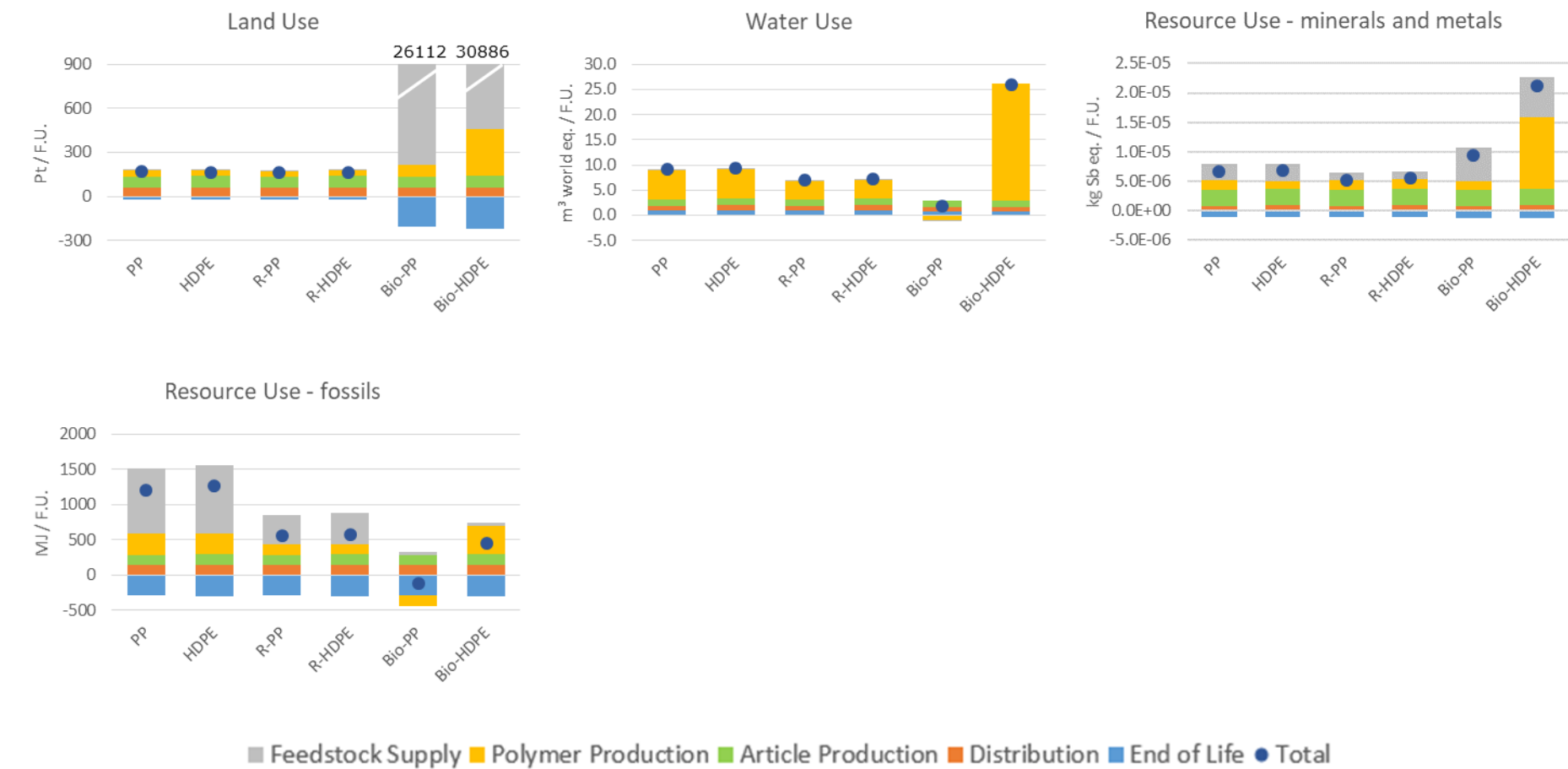
Note that scenario impacts presented in Figures 8.7 to 8.9 refer to the EU-average End of Life scenario (as described in Section 8.4.5.1), and represent the net impacts from waste management activities, resulting from the balance between real burdens and benefits (if any). Potential impacts calculated assuming 100% of post-consumer pots being routed to each viable End of Life option are presented in Figures 8.11 to 8.13, except for Bio-PP and Bio-HDPE pots, which for reasons of scale are separately presented in Annex B.5, Tables B.5.10 and B.5.11. We warn that this analysis should not be interpreted as a direct LCA comparison of alternative End of Life management options for nursery pots, because a product-based perspective is considered and allocations among consecutive product life cycles are applied to the recycling pathway. Conversely, in a typical waste management LCA that compares these alternative End of Life options (i.e. with a functional unit of 1 tonne of waste managed), the recycling pathway would be assigned the full burdens and benefits it involves, as there is no need to break the system mass flows between the system recycling and the one using the recycled feedstock (i.e. system perspective; no allocation needed).



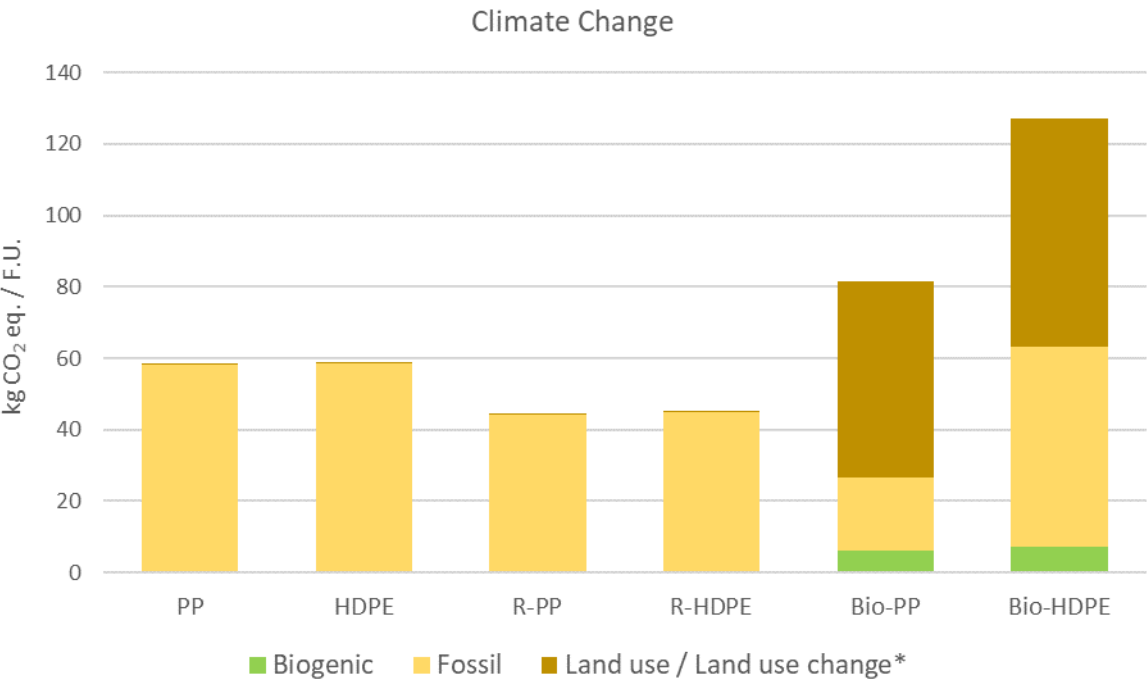
**Figure 8.7.** Potential impact of nursery pots LCA scenarios for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity – non-cancer, Particulate Matter and Ionising Radiation. Note that in some impact categories a part of the results is out of scale and is curtailed. Climate Change impacts denoted with “C bio EoL” accounts for the contribution of biogenic carbon not released after 100 years from landfilling of bio-based pots.



**Figure 8.8.** Potential impact of nursery pots LCA scenarios for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater. Note that in some impact categories a part of the results is out of scale and is curtailed.

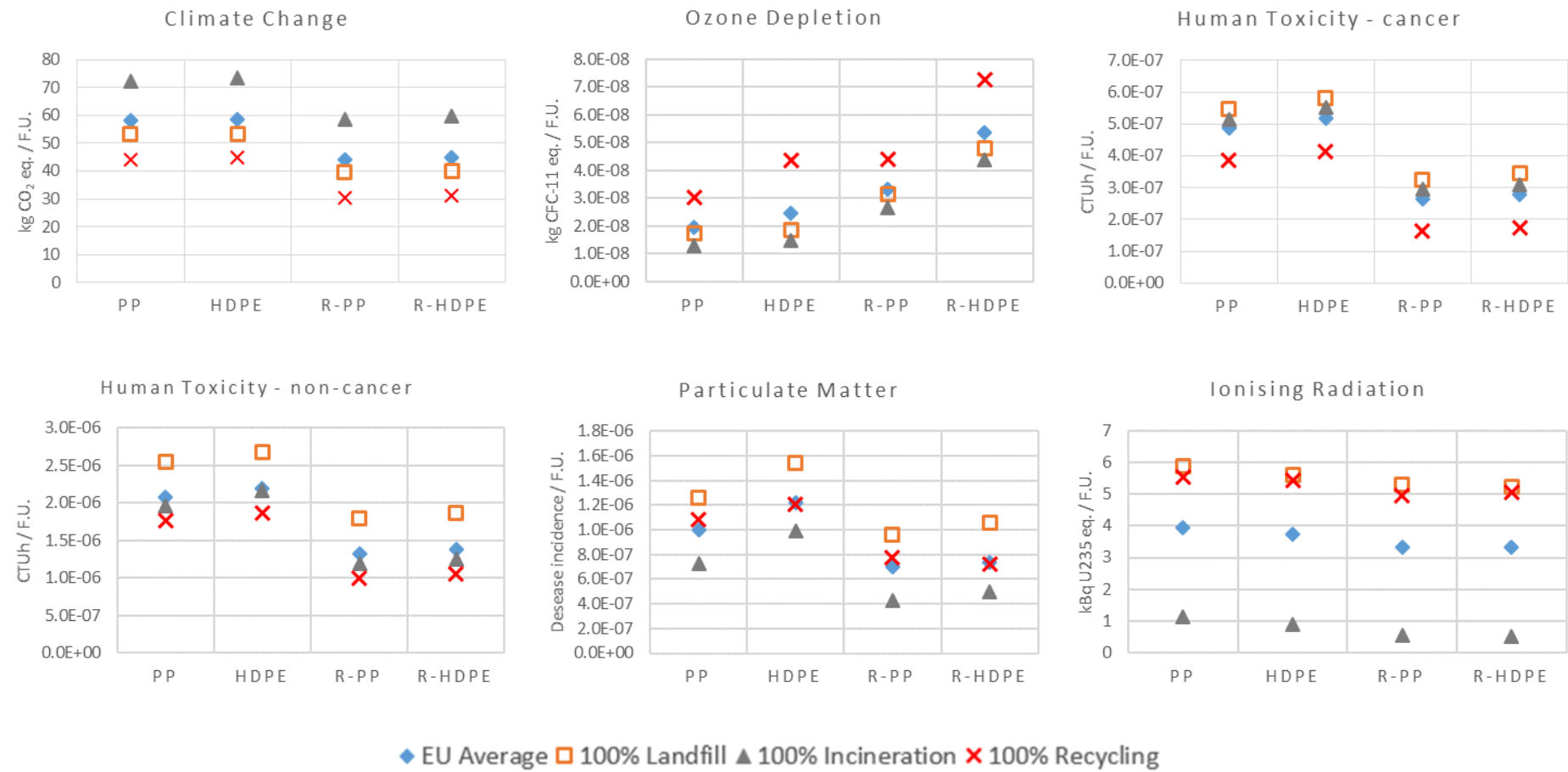


**Figure 8.9.** Potential impact of nursery pots LCA scenarios for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use - fossils. Note that in some impact categories a part of the results is out of scale and is curtailed.



**Figure 8.10.** Contribution of fossil, biogenic, and land use/land transformation GHG emissions to the total Climate Change impact of nursery pots LCA scenarios (note that the contribution of iLUC to land transformation emissions is excluded).

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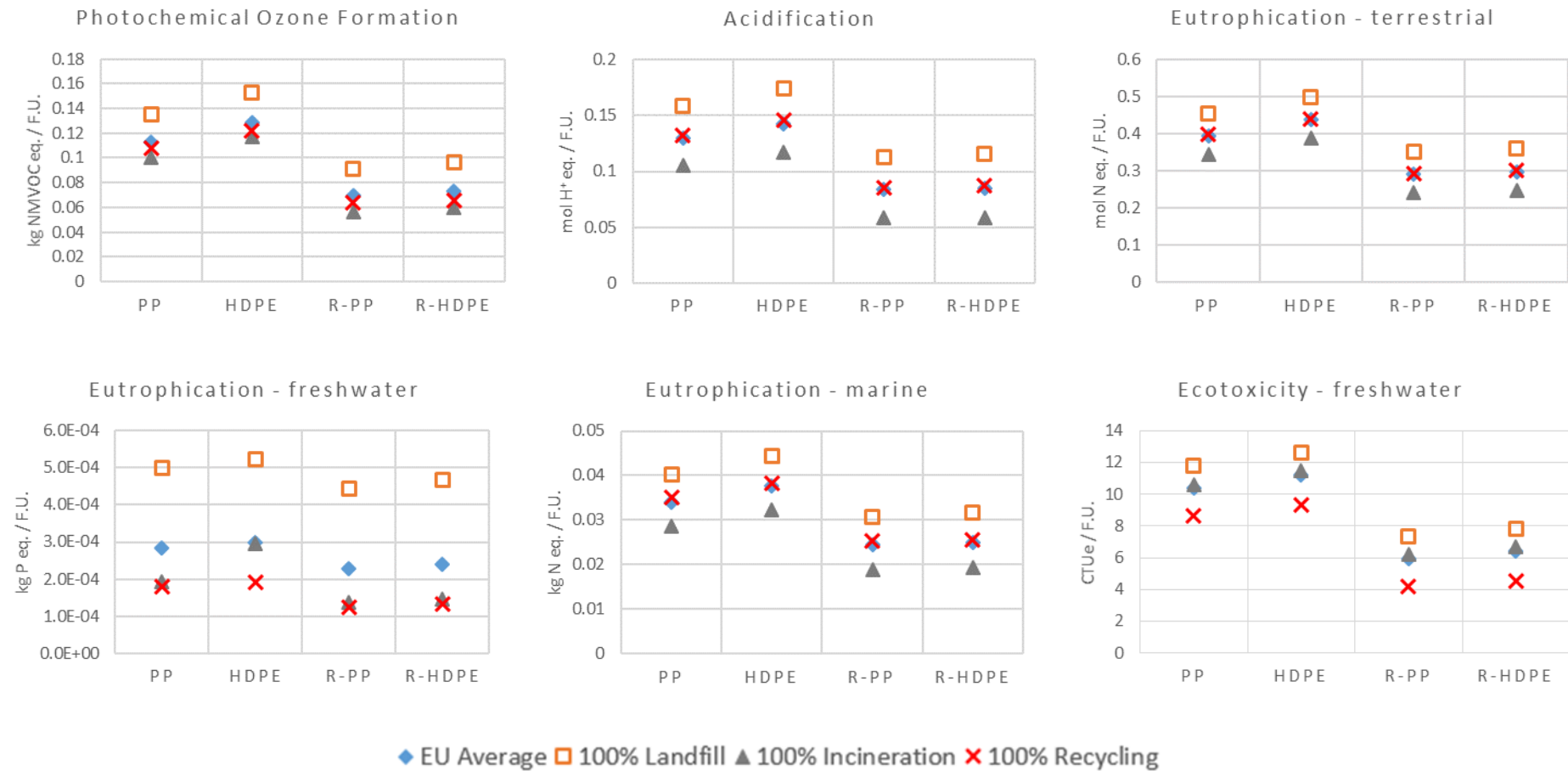
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**Figure 8.11.** Potential impact of nursery pots LCA scenarios for different End of Life options, for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity - non-cancer, Particulate Matter and Ionising Radiation.

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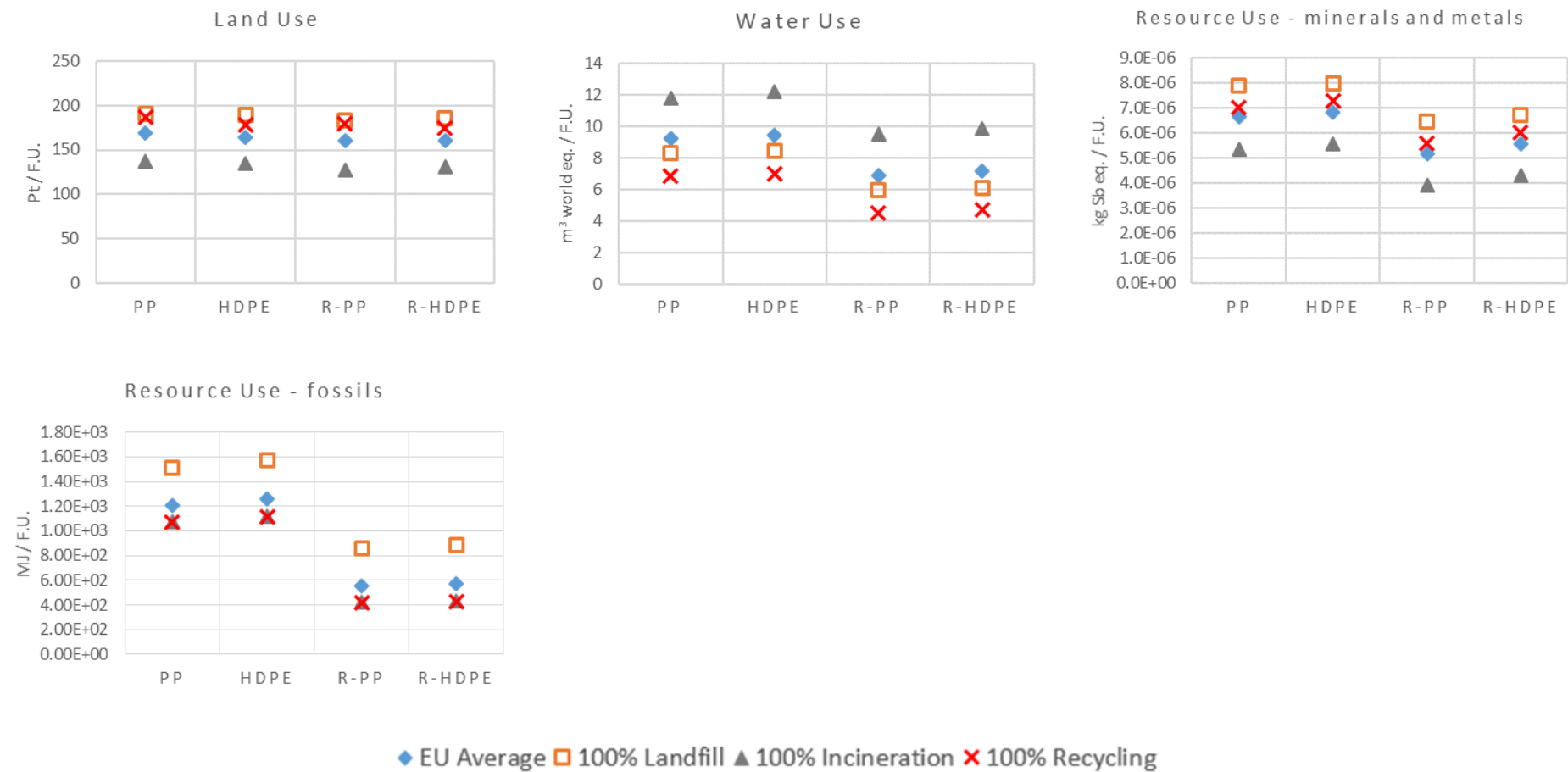


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**Figure 8.12.** Potential impact of nursery pots LCA scenarios for different End of Life options, for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater.



**Figure 8.13.** Potential impact of nursery pots LCA scenarios for different End of Life options, for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils.



## 8.6 Additional Environmental Information

This section presents additional environmental impacts or aspects going beyond the default impact categories considered in the present method, which reflect official categories adopted in the PEF context. Additional environmental issues addressed include indirect Land Use Change (iLUC) impact on Climate Change, potential Biodiversity impacts, as well as the contribution of the investigated nursery pots scenarios to macro-plastics formation at End of Life (including product litter) and to micro-plastics generation throughout the supply chain.

### 8.6.1 iLUC impacts

Table 8.10 presents the estimated contribution of GHG emissions from iLUC induced by bio-based feedstock supply to the potential Climate Change impact of bio-based nursery pots.

**Table 8.10.** iLUC contribution to the potential Climate Change impact of nursery pots LCA scenarios.

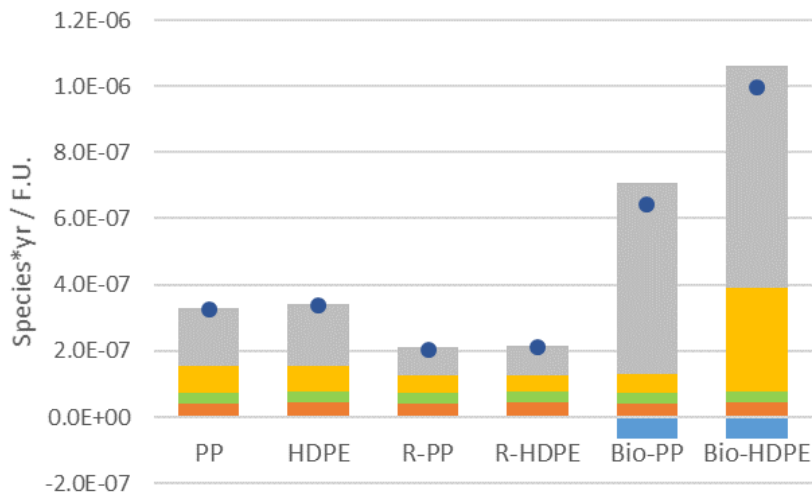
Scenario / Polymer	iLUC Climate Change impact [kg CO <sub>2</sub> eq./FU]	Total Climate Change impact (incl. iLUC) <sup>(1)</sup> [kg CO <sub>2</sub> eq./FU]
S1 – PP	-	(58.1)
S2 – HDPE	-	(58.7)
S3 – R-PP	-	(44.3)
S4 – R-HDPE	-	(45.1)
S5 – Bio-PP	8.91	90.5 (81.6)
S6 – Bio-HDPE	10.4	137.4 (127)

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

### 8.6.2 Biodiversity impacts

Estimated potential Biodiversity impacts of the investigated scenarios for plastic pots are presented in Figure 8.14. It is reminded that the assessment is performed through and endpoint-level indicator relying on impact assessment methods that for some of the underlying midpoint impact drivers differ from those recommended in the present method.

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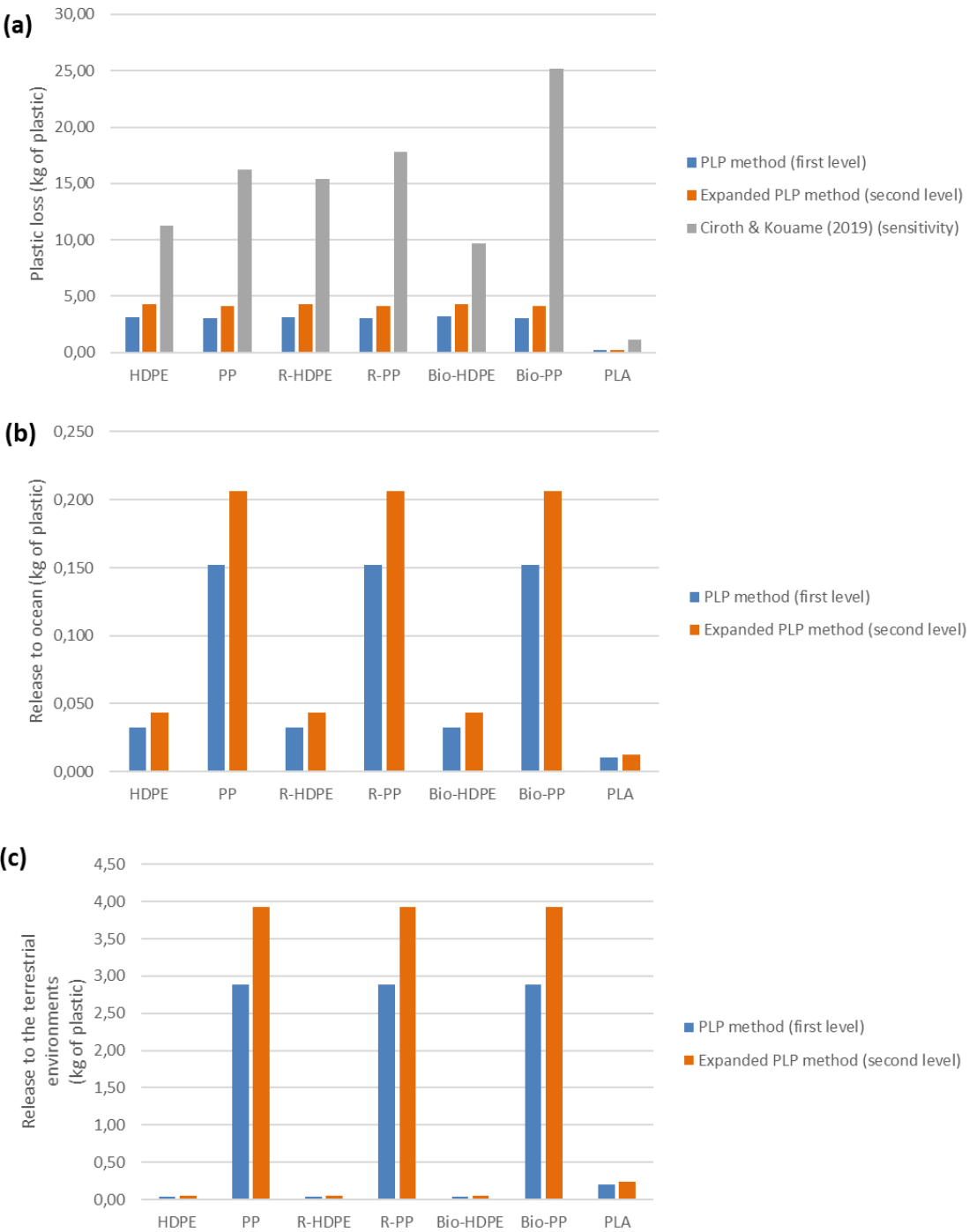


■ Feedstock Supply ■ Polymer Production ■ Article Production ■ Distribution ■ End of Life ● Total

**Figure 8.14.** Potential biodiversity impact of nursery pots LCA scenarios, expressed as potential loss of species per year.

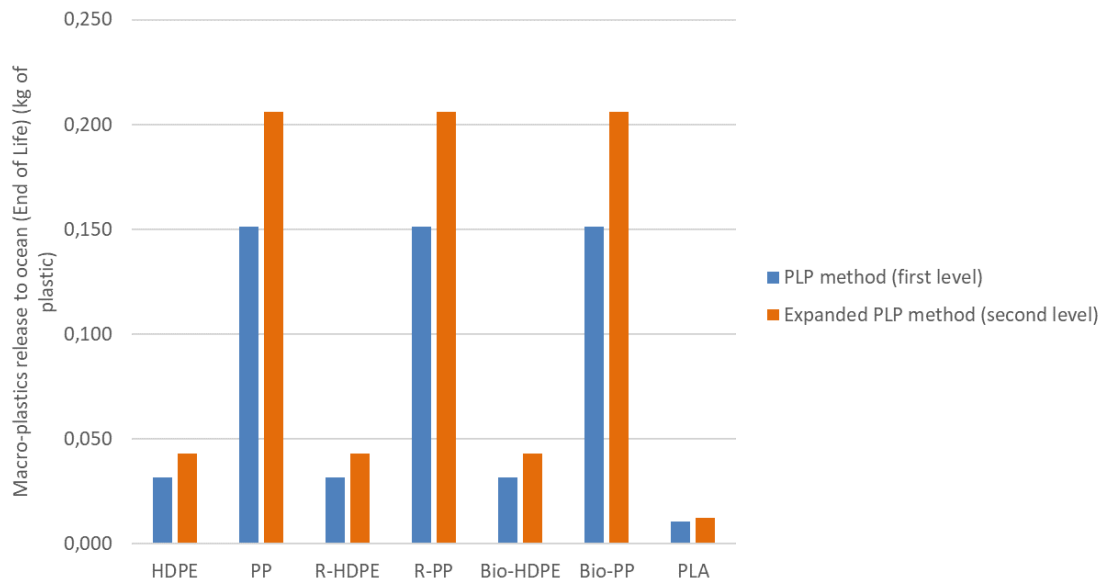
### 8.6.3 Macro- and micro-plastics generation

Figure 8.15 shows the total plastic loss, release to ocean and release to the terrestrial environment estimated for the assessed nursery pot scenarios, considering the whole product life cycle. Estimates obtained by applying all the three different explored approaches are presented, including the total loss and release to ocean/terrestrial environment calculated through both the *PLP method* (first level) and the *Expanded PLP method* (second level), as well as the alternative estimate of the total supply-chain loss of plastics from applying the sensitivity approach by Ciroth and Kouame (2019).



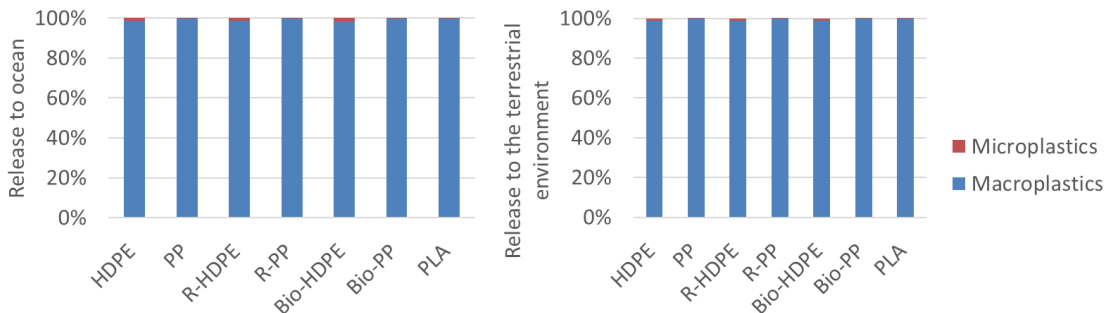
**Figure 8.15.** Total plastic loss (a), release to ocean (b) and release to the terrestrial environment (c) estimated for nursery pots LCA scenarios with different approaches. The contribution of both macro- and micro-plastics is included, considering the whole product life cycle.

Regarding the sole release of macro-plastics to ocean at End of Life (from product littering and waste mismanagement), Figure 8.16 presents the contribution of each nursery pots scenario, estimated with the two different explored approaches: the *PLP method* (first level) and the *Expanded PLP method* (second level).



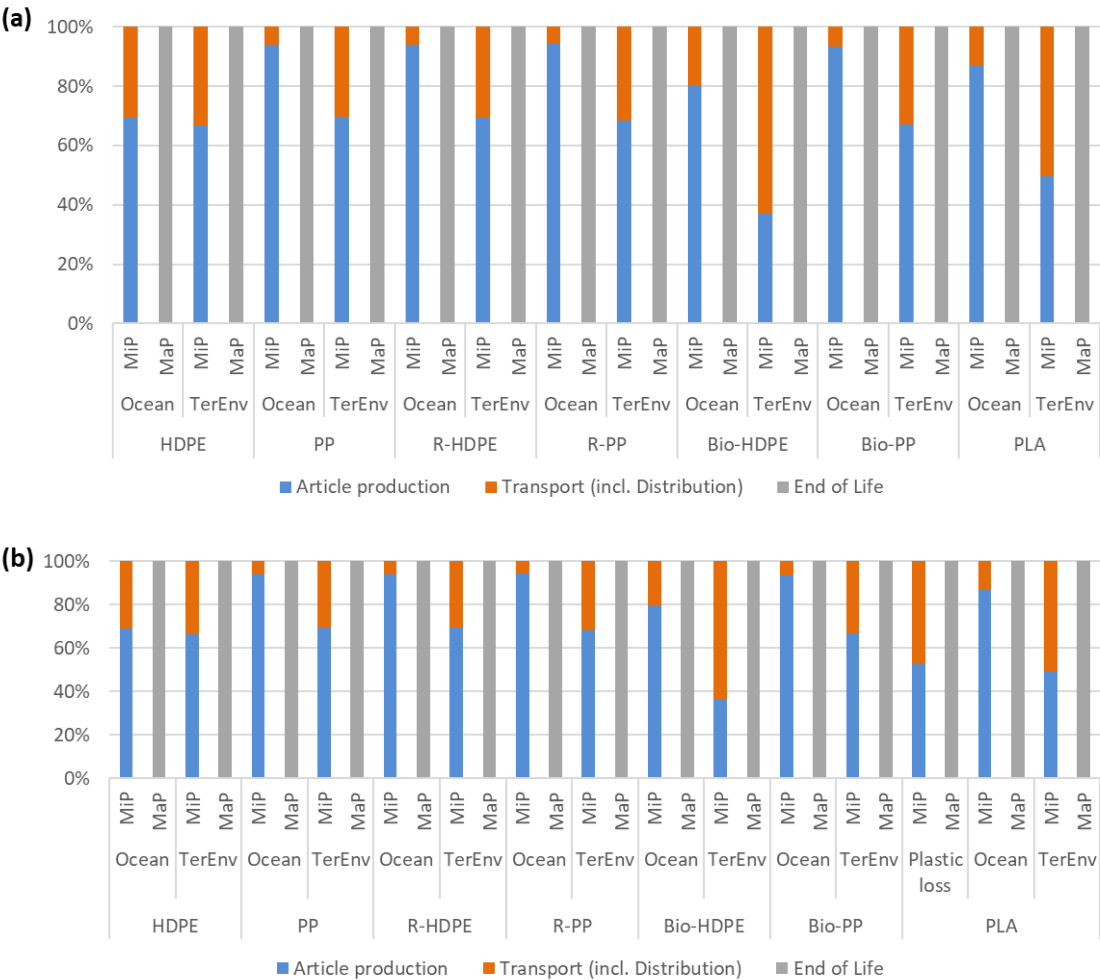
**Figure 8.16.** Macro-plastics release to ocean at End of Life, estimated with different approaches.

As for the share between the release of macro- and micro-plastics, Figure 8.17 shows as an example the relative contribution of the two categories to the total release to ocean and to the terrestrial environment estimated through the *PLP method* (first level). Note that the results are similar to those obtained when applying the *Expanded PLP method* (second level), as further discussed in Section 8.7.7.



**Figure 8.17.** Contribution of macro- and micro-plastics to the total plastic release to ocean and to the terrestrial environment estimated with the PLP method for each nursery pots LCA scenario.

The contribution of the different lifecycle stages to the total macro- and micro-plastics release to ocean and to the terrestrial environment of the assessed nursery pots scenarios has been finally evaluated. The results of this “hotspot analysis” are reported in Figure 8.18 for both the *PLP method* and the *Expanded PLP method*. Results are not shown for the other (sensitivity) approaches as they either cover only plastic losses or only a part of the product life cycle (i.e. End of Life).



**Figure 8.18.** Contribution of life cycle stages to the total macro-plastics (MaP) and micro-plastics (MiP) release to ocean (Ocean) and to the terrestrial environment (TerEnv) estimated for nursery pots LCA scenarios with (a) the PLP method (first level) and (b) the Expanded PLP method (second level).

## 8.7 Interpretation

In the interpretation of case study results, most relevant impact categories and lifecycle stages of the studied systems are firstly identified (8.7.1 and 8.7.2, respectively). Scenario impacts are then compared, considering both characterised LCIA results (8.7.3) and total normalised and weighted impacts (8.7.4). Results obtained by individually applying each viable End of Life option are separately discussed in Section 8.7.6, while the effects of iLUC and the contribution to macro- and micro-plastics generation are addressed in Sections 8.7.5 and 8.7.7, respectively.

Most relevant processes were not identified, since it was not possible to achieve a similar level of vertical disaggregation for foreground processes included in the life cycle models of the analysed scenarios (e.g. a higher disaggregation could be achieved for bio-based HDPE pots, but this was not the case of other scenarios). Therefore the identification of most relevant processes cannot be carried out consistently across all scenarios, and a more detailed investigation for specific scenarios would not be meaningful. The identification of most relevant elementary flows was also not undertaken, as this would require prior identification of most relevant processes. Note, however, that any company, organisation or any other supply chain actor applying the present method shall proceed with the identification of both most relevant processes and elementary flows.

### 8.7.1 Identification of most relevant impact categories

Table 8.11 shows the most relevant impact categories identified for each nursery pots scenario, based on normalised and weighted impacts, according to the approach outlined in the Method (Report I, Section 6.2.1). Relevant categories were identified as those that cumulatively contribute to at least 80% of the total normalised and weighted impact of the considered impact categories, excluding toxicity-related ones (these being still based on characterisation factors implemented in the EF 2.0 impact assessment methods, and hence not yet updated based on REACH data<sup>145</sup>). 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 fossil-based (i.e. PP and HDPE) and recycled PP pots, only two impact categories are identified as the most relevant ones, i.e. Climate Change and Resource Use – fossils. Climate Change is the most relevant for both fossil-based and recycled PP, while for virgin HDPE, Resource Use – fossils is the most relevant one. These two categories are also the most relevant for recycled HDPE, although in this case also Particulate Matter is included, even though with a limited contribution (4.3%).

In the case of bio-based PP and HDPE pots, Particulate Matter is the most relevant category, followed by Climate Change. For bio-based HDPE also Eutrophication – marine is included, although with a small contribution (3.9%).

**Table 8.11.** Most relevant impact categories identified for nursery pots LCA scenarios and related contribution to the total normalised and weighted scenario impact.

S1 - Fossil-based PP		S2 – Fossil-based HDPE		S3- Recycled PP	
Impact category	Contrib.	Impact category	Contrib.	Impact category	Contrib.
Climate Change	41.1%	Resource Use - fossils	40.9%	Climate Change	50.0%
Resource Use – fossils	41.0%	Climate Change	39.9%	Resource Use - fossils	30.2%
Acidification	3.9%	Particulate Matter	4.4%	Particulate Matter	4.1%
<b>Total</b>	<b>86.1%</b>	<b>Total</b>	<b>85.3%</b>	<b>Total</b>	<b>84.4%</b>
S4 – Recycled HDPE		S6 – Bio-based PP		S6 – Bio-based HDPE	
Impact category	Contrib.	Impact category	Contrib.	Impact category	Contrib.
Climate Change	49.9%	Particulate Matter	68.5%	Particulate Matter	64.8%
Resource Use - fossils	30.1%	Climate Change	12.3%	Climate Change	14.1%
Particulate Matter	4.3%	Eutrophication - marine	4.4%	Eutrophication - marine	3.9%
<b>Total</b>	<b>84.4%</b>	<b>Total</b>	<b>85.3%</b>	<b>Total</b>	<b>82.8%</b>

<sup>145</sup> According to the latest version of the Product Environmental Footprint Category Rules Guidance (EC, 2018), toxicity-related impact indicators based on these characterization factors shall be excluded from the procedure to identify most relevant impact categories.

## 8.7.2 Identification of most relevant life-cycle stages

Table 8.12 shows the most relevant lifecycle stages in the relevant impact categories identified in Section 8.7.1, and the associated contribution, quantified according to the rules detailed in the Method (Report I, Section 6.2.2). Most relevant stages include those that together contribute to at least 80% of the total characterised lifecycle impact in the specific category, and are highlighted in yellow in Table 8.12. For a more exhaustive picture, the contribution of all lifecycle stages is reported.

For fossil-based PP and HDPE pots (both virgin and recycled), Polymer Production is identified as the most relevant stages, in terms of Climate Change. It is followed by Feedstock supply, Distribution and Article Production stages (for virgin options) and by Distribution, Article Production and End of Life (for recycled options). Life-cycle stages contributions are similar between virgin polymers and between recycled polymers. In terms of Resource Use - fossils, feedstock supply appears as the most relevant life-cycle stage, covering alone to more than 70% of the total impact of that category. It is followed by Polymer Production, which is substituted by Article Production for recycled HDPE. In terms of Particulate matter, relevant only for R-HDPE pots, the most relevant stages are Distribution and Article Production whose contribution is 40% each.

For bio-based PP and HDPE pots, Feedstock Supply is the most relevant stage in all the impact categories identified as most impacted. In bio-based PP pots, it contributes, alone to more than 90% of the overall impact, in terms of Particulate Matter. This is likely associated with relevant particulate emissions from the pre-harvest burning practice applied to 45% of Brazilian sugarcane.

A similar situation is observed also for bio-based HDPE pots, where Feedstock Supply is alone responsible for 73-89% of the total impact in all of the tree relevant categories, reflecting the relatively high specific consumption of sugarcane for polymer production (28.6 kg of sugarcane per kg of HDPE). For Climate Change and Eutrophication – marine, Feedstock

**Table 8.12.** Contribution of life cycle stages to the total characterised impacts of the most relevant categories identified for nursery pots LCA scenarios. Most relevant stages (i.e. those contributing to more than 80% of the total impact) are highlighted in yellow.

S1 – Fossil-based PP					
Climate Change		Resource Use - fossils		Acidification	
Life cycle stage	Contrib.	Life cycle stage	Contrib.	Life cycle stage	Contrib.
Polymer Production	38.7%	Feedstock Supply	75.8%	Polymer Production	37.6%
Feedstock Supply	18.8%	Polymer Production	25.6%	Feedstock Supply	33.7%
Distribution	18.2%	Article Production	11.7%	Distribution	28.0%
Article Production	15.1%	Distribution	11.4%	Article Production	20.8%
End of Life	9.2%	End of Life	-24.5%	End of Life	-20.2%

1

S2 – Fossil-based HDPE					
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	77.5%	Polymer Production	35.9%	Polymer Production	50.4%
Polymer Production	23.6%	Feedstock Supply	19.7%	Feedstock Supply	24.7%
Article Production	11.8%	Distribution	18.7%	Distribution	24.2%
Distribution	11.4%	Article Production	15.7%	Article Production	23.8%
End of Life	-24.2%	End of Life	9.9%	End of Life	-23.1%
S3 – Recycled PP					
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	32.6%	Feedstock Supply	74.5%	Distribution	40.5%
Distribution	23.8%	Polymer Production	28.6%	Article Production	40.0%
Article Production	19.9%	Article Production	25.3%	Polymer Production	30.8%
End of Life	12.1%	Distribution	24.5%	Feedstock Supply	21.2%
Feedstock Supply	11.6%	End of Life	-52.9%	End of Life	-32.5%
S4- Recycled HDPE					
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	30.2%	Feedstock Supply	77.2%	Distribution	40.0%
Distribution	24.4%	Article Production	26.0%	Article Production	39.3%
Article Production	20.5%	Polymer Production	25.1%	Polymer Production	37.6%
End of Life	12.9%	Distribution	25.0%	Feedstock Supply	21.3%
Feedstock Supply	12.1%	End of Life	-53.2%	End of Life	-38.1%
S5 – Bio-based PP					
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	97.6%	Feedstock Supply	89.0%	Feedstock Supply	90.1%
Polymer Production	2.1%	Distribution	11.8%	Polymer Production	8.4%
Distribution	0.3%	Article Production	9.9%	Distribution	1.7%
Article Production	0.3%	Polymer Production	4.3%	Article Production	0.7%
End of Life	-0.4%	End of Life	-14.9%	End of Life	-0.9%

2

3



<b>S6 – Bio-based HDPE</b>					
<b>Particulate Matter</b>		<b>Climate Change</b>		<b>Eutrophication - marine</b>	
<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	88.8%	Feedstock Supply	72.9%	Feedstock Supply	87.3%
Polymer Production	11.0%	Polymer Production	21.9%	Polymer Production	11.4%
Distribution	0.3%	Distribution	8.7%	Distribution	1.5%
Article Production	0.3%	Article Production	7.3%	Article Production	0.6%
End of Life	-0.3%	End of Life	-10.8%	End of Life	-0.8%

### 8.7.3 Interpretation of characterised results

In this section, the characterised potential impacts of the assessed scenarios (Figures 8.7-8.9 and Table B.5.7) are compared to evaluate the effects of using an alternative feedstock or material for a specific fossil-based polymer of reference used in nursery pots manufacturing. Therefore, the impacts of recycled PP and bio-based PP are compared with those of virgin fossil-based PP pots, while recycled HDPE and bio-based HDPE pots are compared with fossil-based HDPE pots. In addition, the impacts of reference scenarios (fossil-based PP and HDPE pots) are initially compared, to provide useful elements to potentially extend the comparative considerations provided below to any of such reference scenarios.

The comparison focuses on the sixteen default impact categories considered in the Product Environmental Footprint (PEF) context and in this assessment. Potential impacts on Biodiversity are discussed separately, as being estimated via an endpoint indicator that is not recommended in the PEF framework, and partially relying on different impact assessment methods for the underlying midpoint-level impact categories. Differences between scenario impacts lower than 10% were not considered significant, in light of the uncertainty associated with the developed life cycle models and the applied impact assessment models. Moreover, any comparative considerations related to Human Toxicity and Ecotoxicity impact categories need to be interpreted in light of the higher uncertainty of the underlying impact assessment models and results. The same applies, to a lower extent, also to Water Use and Land Use.

Focusing on the relative performance of pots made of the two fossil-based reference materials (i.e. PP and HDPE), none is found to be environmentally superior to the other. In fact, PP pots are preferable in 5 out of 16 impact categories (including Ozone Depletion, Eutrophication – terrestrial and marine, Ecotoxicity – freshwater, and Resource Use – fossils), mainly thanks to the lower impact from Polymer and Article Production stages. This reduction compensate for the usually increased impact from Feedstock Supply, and also End of Life in the case of Ozone Depletion. On the other hand, HDPE pots are preferable only in 2 impact categories, namely Ionizing radiation and Land Use, and mostly due to the lower impact from Polymer Production. For the remaining categories, which are the majority (9 over 16), pots made of both polymers show a comparable performance (i.e. relative impact variation lower than 10%).

The use of 100% recycled PP and HDPE in pots manufacturing improves the environmental performance compared to their virgin counterparts in all impact categories. The only exception is Ozone Depletion, where pots made of both recycled polymers show a worsened performance, especially when R-HDPE is used (+116%), although also for R-PP the impact increase is significant (+72%). The explanation of this can be found in the emissions of ozone depleting substances from the life cycle of sodium hydroxide used in secondary PP and HDPE production (which is responsible for more than

90% of the increased polymer production impacts). Leaving this exception apart, the categories that improve the most are Resource use –fossil, Human Toxicity – cancer and Ecotoxicity – freshwater, where the impact is reduced by 54%, 46% and 43% respectively for R-PP pots. Similar reductions are also observed for R-HDPE pots. Such relevant improvements reflect the high recycled material content, assumed to be 100% in these scenarios, in the absence of representative data on the current average recycled content in pots at the EU level (Section 8.1).

The use of sugarcane-derived bio-polymers (i.e. Bio-PP and Bio-HDPE) substantially increases the impacts of the pot life cycle compared to the use of their fossil-based counterparts. An exception is Resource Use – fossils, which is decreased (110% for bio-PP pots and 65% for bio-HDPE pots) thanks to the reduced use of fossil feedstock sources. In the case of bio-PP pots, also Ozone Depletion is an exception, showing a decrease of 73%. The main responsible for this overall worsened performance is the increased feedstock supply impact, which (also) reflect the high specific sugarcane consumption for bio-PP and bio-HDPE production (equalling 25.7 kg per kg of PP and 28.6 kg per kg of HDPE). Concerning the detailed analysis of the impact increase, category by category, the situation observed for bio-PP pots is similar to the one for bio-HDPE pots, as described below. The highest increases are observed for Land Use (due to increased land demand for biomass growth), Particulate Matter (due to the pre-harvest burning practice in sugarcane cultivation), and for both marine and freshwater Eutrophication (due to nutrient emissions from cultivation). Relevant increases also take place in Human toxicity – non cancer, Eutrophication – terrestrial (due, again, to nutrient emissions), Photochemical ozone formation (CO and NO<sub>x</sub> emissions), and Acidification (NO<sub>x</sub> emissions), where also increased transport impact of bioethanol from Brazil to Europe plays a role (in the case of Bio-HDPE) beyond emissions from sugarcane cultivation. For the remaining categories, the increase is lower, but still significant, ranging from 20% (Ionising Radiation) to 112% (Human toxicity – cancer), for Bio-PP, and from 27% (Ionising Radiation) to 284% (Ozone Depletion), in the case of Bio-HDPE. Note that in the case of Ozone Depletion, Ionising Radiation, and Water Use, the main responsible for the worsened performance is the increased impact of Polymer Production, rather than of Feedstock Supply. In particular, in the case of Bio-HDPE<sup>146</sup>, Ozone Depletion impacts are dominated by chemicals and natural gas used during polymerisation, while production of electricity used in the same process dominates Ionising Radiation impacts. For Water Use, the largest contribution is provided by water consumption in Ethanol production. In this respect, it is highlighted that the values of water consumption considered in the adjusted Ecoinvent dataset used to model Ethanol production are higher than those reported in the inventory developed from Braskem (ACV Brasil, 2017). For consistency reasons, no changes were made to default values.

Focusing on estimated potential impacts on biodiversity (Figure 8.14), both recycled PP and recycled HDPE pots shows a reduced impact (-40%) compared to pots made of their virgin counterpart. Conversely, bio-based PP and bio-based HDPE pots involve a higher impact, especially in the case of Bio-HDPE. This is due to the increased impact from feedstock production, and the resulting land occupation for sugarcane growing. In the case of Bio-HDPE, also an increased Polymer Production impact is observed (and dominated by the impact of biogenic CO<sub>2</sub> emissions from sugarcane conversion to ethanol). It has to be reminded, however, that direct potential biodiversity (ecosystem) impacts from oil leakage are not quantified for pots made of fossil-based polymers considered as a reference (although emissions from leakage per unit of oil supplied are reported to be quite small; see Section 4.4.1.1).

As a last comment, it is noted how the Climate Change impact of bio-based pots (i.e. Bio-PP and Bio-HDPE pots) is moderately reduced if the contribution of the missed release of biogenic carbon after 100 years from landfilling is taken into account. Indeed, a decrease between 12% (Bio-PP) and 18% (Bio-HDPE) is observed. This is partly explained by the

<sup>146</sup> For Bio-PP no further investigation was possible due to the aggregated nature of the applied gate-to-gate inventory dataset.

limited share of landfilling in the applied EU-average scenario (31%), despite the 100% biogenic carbon content in the polymer.

#### **8.7.4 Interpretation of normalised and weighted results**

This section briefly compares the investigated scenarios based on the respective total impact scores, calculated by aggregating normalised and weighted impact assessment results across all impact categories (Table B.5.9, Annex B.5). The comparison is carried out according to the same criteria outlined in Section 8.7.3 in terms of contrasted nursery pots scenarios, and considering significant only impact differences larger than 10%. While affected by greater uncertainty (from the calculation and application of normalisation and weighting factors), total impact scores provides a more immediate and synthetic indication of the overall environmental performance of the analysed product scenarios. This facilitates the interpretation compared to looking at characterised results from several impact categories. However, such estimates are inevitably affected by value choices underlying the definition of weighting factors aimed at establishing an order of relevance for the different impact categories in a European decision context.

The comparison among total normalised and weighted scenario impacts reflects the overall picture of results emerged when focusing on characterised impact assessment results in single impact categories, and can be summarised in the following points:

- Recycled PP and HDPE pots show an improved performance compared to their fossil-based counterparts, halving the total impact score.
- Bio-based pots show a substantially worse performance compared to their fossil-based counterpart (both virgin and recycled), with bio-based HDPE pots being the worst performing option among those assessed. They show a total impact score that is six times higher than virgin HDPE and ten times higher than recycled HDPE.

#### **8.7.5 Effects of indirect land use change (iLUC)**

When the contribution of GHG emissions from iLUC is taken into account, only moderate changes are identified in the Climate Change impact of bio-based alternatives (i.e. Bio-PP and Bio-HDPE pots) (Figure 8.7 and Table 8.10). A maximum increase of 11% is observed for bio-based HDPE pots, while for bio-based PP pots the iLUC contribution is even lower (8%), despite being the two alternative options fully bio-based. This is partially a consequence of the higher Climate Change impact associated with the use of these materials even without considering iLUC.

Overall, accounting for iLUC effects does not affect the relative performance of the assessed scenarios (i.e. their comparison is not affected). However, this outcome needs to be read also in light of the GHG emission factors applied for iLUC in this assessment, which appear to fall in the lower end of the range of values available in the recent literature (see Section 4.4.16.3 of the Method (Report I)). Moreover, the iLUC contribution considered in this study only accounts for GHG emissions, so only impacts on Climate Change were addressed. A larger iLUC GHG contribution and the inclusion of nutrients-related impacts (due to e.g. intensification) may (further) worsen the Climate Change, Acidification, and Eutrophication performances of the two bio-based pots scenarios.

#### **8.7.6 Alternative End of Life options**

This section discusses characterised scenario results calculated by individually applying each viable End of Life option as an exclusive (100%) End of Life scenario (as presented in Figures 8.11 to 8.13 and in Annex B.5 for what concerns bio-based scenarios). The main purpose is to evaluate how the performances of single nursery pots scenarios are affected by the change in the applied End of Life option, rather than to mutually compare such scenarios for different End of Life options. However, the results should not be

interpreted as a strict comparison among different End of Life alternatives for plastic pots, since the evaluation applies a product perspective, and burdens/benefits of selected End of Life options (e.g. recycling) are partitioned between different product life cycles (which prevents from capturing the full implications of having a given waste stream routed to a specific End of Life option). In a waste management system perspective, where allocation is not needed, the total (system-wise) savings associated with the End of Life pathway “100% recycling” as such would likely be higher than what presented in this report where a product-perspective is applied.

None of the three considered product End of Life scenarios (i.e. mechanical recycling, incineration and landfilling) can be identified as preferable across all the assessed impact categories and nursery pots scenarios. For pots made out of fossil based polymers, both virgin and recycled, mechanical recycling is the preferred scenario in seven categories, including Climate Change, Human Toxicity – cancer, Human Toxicity – non-cancer, Eutrophication – freshwater, Ecotoxicity – freshwater, Water Use and Resource Use – fossils. For bio-PP and bio-HDPE pots, a similar situation can be observed, with the exception of Climate Change.

In Ozone Depletion, mechanical recycling shows the worst performance for all the assessed scenarios, including bio-based ones. In Ionising Radiation, recycling is almost comparable with landfilling, which is the least preferable scenario. For bio-PP and bio-HDPE pots, this is also the case of Land Use.

Incineration is the most favourable scenario in most of the categories for all the assessed scenarios. This is likely a consequence of the avoided emissions from electricity substitution. For fossil-based polymers (i.e. PP and HDPE, virgin and recycled) incineration is the preferred End of Life scenario in nine categories over the sixteen considered (i.e. Ozone Depletion, Particulate Matter, Ionising Radiation, Photochemical Ozone Formation, Acidification, Eutrophication – terrestrial, freshwater and marine, Land Use and Resource Use – minerals and metals), while for bio-based polymers only in terms of Human Toxicity –cancer, Ecotoxicity – freshwater and Water use incineration is not the best option, as explained below.

In many impact categories, landfilling is the worst scenario for PP and HDPE nursery pots, in line with the priority order outlined in the “Waste Hierarchy”, which sets disposal as the least preferable option (EC, 2008). Exceptions are Climate Change, Ozone Depletion, and Water Use, where other scenarios show the highest impact, as discussed above. For Ionising radiation and Land Use, landfilling is essentially comparable with the other scenarios, although still showing the worst performance within the limited range of impacts. Landfilling is never preferable in the case of fossil-based PP and HDPE pots, both from virgin and recycled feedstock. For bio-based PP and HDPE pots, landfilling is the less favourable scenario for all the impact categories, with the exception of Ozone Depletion and Water Use, where the worst option is represented by recycling and incineration respectively. However, if the contribution of non-released biogenic carbon after 100 years from landfilling is taken into account, the 100% landfilling scenario becomes preferable for both bio-based pots.

#### **8.7.7 Interpretation of results related to macro- and micro-plastics generation**

This section discusses the results presented in Section 8.6.3 on the estimated potential generation of macro- and micro-plastics of the assessed nursery pots scenarios. The latter are compared according to the same criteria applied to the interpretation of the potential environmental impacts (Section 8.7.3), initially focusing on the results from the two-level approach applied as a base case. The results from the approaches applied as a sensitivity analysis are discussed separately.

Regarding the total loss of plastics (macro- and micro-plastics) and the resulting release to ocean and to the terrestrial environment (Figure 8.15), the two base-case approaches (*PLP method* and *Expanded PLP method*) provide similar results when comparing the

different nursery pots scenarios. In both cases, the scenario considering the use of PLA as alternative feedstock sources for HDPE and PP pots show a significantly lower plastic loss (-93%) and release to ocean (-33%) when compared to HDPE. This is mostly a consequence of the amount of polymer required per functional unit for the different scenarios, which leads to a proportionally reduced or increased loss and release of plastics from both upstream life cycle stages (micro-plastics) and especially from product End of Life (macro-plastics). Note, however, that the applied methods rely on mass-based indicators (which are affected by differences in pot mass) rather than quantifying the number of items (or plastic fragments/particles) lost or released to the environment. If this was the case, all scenarios would result in the same macro-plastics generation at End of Life when applying the *PLP method*, since pots with a lower mass would not imply a reduction in the contribution of product littering and waste mismanagement (which are based on release rates that are not affected by the type of feedstock or material). Conversely, HDPE-based scenarios (fossil-based, recycled and bio-based) show the lowest release to the terrestrial environment among the assessed scenarios. This is a consequence of the modelling parameters, where release rates to the ocean and the terrestrial environment are set as 1% for HDPE due to its economic value and the assumption that informal waste picking prevents the release to the environment.

Comparing the two base-case approaches, the *Expanded PLP* method results in a 36% higher plastic loss and release to ocean and the terrestrial environment compared to the original *PLP method* (Figure 8.15). This is mainly due to an increased contribution of macro-plastics loss at End of Life, which also accounts for mismanagement of plastic waste exported for (sub-standard) recycling outside the EU (where the overall waste mismanagement share is increased by 2-4% compared to the assumption that all plastic waste is recycled in Europe). Conversely, the two additional micro-plastics sources considered in this alternative approach (road markings and marine coatings) does not significantly affect the results, due to the overall modest contribution of micro-plastics to the total release (in terms of mass).

The total supply-chain loss of plastics estimated based on the approach by Ciroth and Kouame (2019; sensitivity analysis) is between 3 and 5.8 times higher compared to applying the *PLP method*, with an even higher increase for Bio-PP pots (8.3 times). This is likely a consequence of the broader range of processes covered (albeit in a more generic manner) compared to the *PLP method* (e.g. the contribution of background transport processes is also accounted for). Comparative results among the different nursery pots scenarios are similar to those obtained in the base-case assessment, with PLA pots showing the lowest total loss<sup>147</sup>, and with no relevant differences when changing the type of feedstock used for PET or PP pots production. An exception is represented by Bio-PP pots, which show a larger plastic loss compared to their completely fossil-based counterpart (+160%). However, this result is affected by the approximations performed in the modelling of Bio-PP production for plastic loss calculation, and may not be sufficiently reliable. More in general, the results obtained by applying this alternative approach need to be interpreted with caution, since the life cycle models of some nursery pots scenarios present different level of (vertical) disaggregation, and the contribution from intermediate or background processes may have not been taken into account consistently. Moreover, approximations in the modelling of the production of bio-based polymers (beyond the case of Bio-PP discussed above) also affect the respective contribution.

Focusing on the relation between total loss and release (which depends on the parameters of the *PLP method*), this depends on the polymer. Regarding PP and PLA, the plastic loss is modelled as fully released to the environment due to the lack of informal waste collection for this type of product. In relative terms, release to ocean accounts for a lower share of the plastic loss (5%), while release to the terrestrial environment is the most relevant (95%). Regarding HDPE, the higher economic value leads to release rates

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<sup>147</sup> The same comments as above on the implications of using a mass-based indicator need to be taken into account in the interpretation of this result.

of 1%. As shown in Figure 8.17 for the *PLP method*, the total plastic release to both ocean and the terrestrial environment is dominated by the release of macro-plastics at the End of Life (more than 99%), while micro-plastics released throughout the upstream life cycle have only a minor role. Similar results are also observed for the *Expanded PLP method* (not shown), despite two additional sources of micro-plastics are considered in the latter (i.e. road markings and marine coatings). In both cases, this is a consequence of the prevailing mass of macro-plastics, which is directly related to the mass of pots required per functional unit (reference flow), by means of higher loss and release rates compared to micro-plastics. Therefore, the mass of macro-plastics is at least one order of magnitude higher than the mass of micro-plastics, which instead depends on the quantity of relevant lifecycle processes by means of (much) lower loss and release coefficients (depending on the source).

Due to the prevailing role of macro-plastics, the comparison among the sole macro-plastics release to ocean (from product littering and waste mismanagement) estimated for nursery pots scenarios with both base-case approaches reflects the results obtained when looking at the total plastic release into such compartment (Figure 8.16). Due to the absence of littering, the release to ocean of macro- and micro-plastics is a consequence of waste mismanagement. Both approaches result in the use of PLA as the scenario with the lowest release. However, it must be reminded that the approaches rely on mass-based indicators, and the resulting trends are thus determined by the mass of material required to fulfil the functional unit. This implies that scenarios relying on lighter pots provide a lower contribution to macro-plastics generation (and vice versa), which would not be the case if an indicator based on the number of released items was considered. Comparing the different approaches, the total release estimated with the *Expanded PLP method* is higher than the estimates from the official *PLP method* (and the highest in general) due to the additional (but more uncertain) contribution from mismanagement of plastic waste exported for recycling outside the EU.

With regard to the origin of released plastics (in terms of life cycle stage and source), macro-plastics entirely originate from the End of Life stage (Figure 8.18), as they are only derived from product littering or mismanagement of the product as waste. On micro-plastics release, the relative contribution of the different sources (e.g. loss of plastic pellets and tire abrasion) depends on the characteristics of the supply-chain (e.g. feedstock origin, which affect the contribution from transport across the life cycle), as well as on the mass of polymer required per functional unit (which affect, to different extents, all the considered sources). However, within the limited role of micro-plastics in the overall release to the environment, the most relevant contribution is generally provided by pellet losses from the Polymer and Article Production stages, while transport-related sources have a more restricted role. An exception is Bio-HDPE pots, where transport emissions are more important due to increased transport throughout the supply chain (although a more disaggregated modelling of the life cycle also affects the relevance of this contribution).

In conclusion, due to the dominant role of macro-plastics loss and release at the End of Life of nursery pots, a best overall performance is associated to the scenario requiring a lower amount of plastic to fulfil the functional unit (i.e. relying on lighter pots; mass-driven results). In this context, the use of PLA pots is the scenario with the lowest plastic loss and release to ocean. However, associated to the modelling of the release rates, the use of HDPE-based (HDPE, recycled HDPE and bio-HDPE) yields the lowest release to the terrestrial environment. The same considerations apply to the sole release of macro-plastics to ocean at End of Life, although in this case the use of an indicator based on the number of items (i.e. pots) released would provide the same result regardless of the scenario when applying the *PLP method* or the sensitivity estimates based on marine litter rates. Pots with different masses would indeed equally contribute to littering and waste mismanagement at End of Life. Conversely, an items-based indicator would still show different results for HDPE pots in the *Expanded PLP method*, with a higher release of macro-plastics due to mismanagement of waste exported for recycling in non-EU countries. From a methodological perspective, the different approaches provide similar

- 1 results when comparing alternative scenarios, apart from few (relevant) discrepancies
- 2 when the total loss of plastics is estimated with the approach by Ciroth and Kouame
- 3 (2019), although the results obtained by applying the latter to the present case study are
- 4 less reliable than the other approaches.

## **9 Case study 6: Insulation boards**

This case study focuses on insulation boards for buildings. These were chosen because they incur a significant demand for plastic feedstock and because they are an example of long-lifetime products. Currently, the European market for building fossil-based insulation materials (i.e. excluding mineral materials such as glass and stone wool) is dominated by expanded polystyrene (EPS) and polyurethane (PUR) which together represent about 83% of the fossil-based polymers supplied to this market sector (see figures elaborated in Pavel and Blagoeva, 2018). Insulation boards made out of these two polymers are considered as a reference of comparison against boards produced from alternative plastic sources, i.e. recycled plastic feedstock, carbon dioxide and biomass.

### **9.1 Assessed scenarios**

The use of different materials and/or feedstock for manufacturing of insulation boards was explored by assessing a number of alternative scenarios (Table 9.2). Two reference scenarios based on fossil-based plastics (100% virgin PUR and EPS made of 98% virgin material) were analysed. The use of recycled fossil-based post-consumer plastic waste as a feedstock (recycled EPS and PET; R-EPS and R-PET) was also explored (Scenario 3-4), assuming a 100% recycled content. Although different shares of recycled material can be mixed with virgin material to be used as input for the production of insulation boards, this study focuses on articles relying entirely on recycled input as this is already in practice (e.g. see R-PET boards by Armaform and R-EPS products by eps360®). This allows assessing the effects of a complete substitution of the virgin material. The use of CO<sub>2</sub> captured from point emission sources as a feedstock for polyols used in PUR production was explored in scenario 5. Capture from coal-fired power plants (baseline) and ammonia producing plants (in a dedicated sensitivity analysis) was specifically considered. Captured CO<sub>2</sub> is then used in partial substitution of propylene oxide as ingredient to produce polyols, following the process described in Fernandez-Dacosta et al. (2018). Due to the innovative nature of this alternative route, the LCI data used in the modelling mostly derive from process simulations and modelling exercise found in the literature, rather than full-scale production facilities. While possible process optimisation strategies may have been taken into account in the generation of such data (e.g. energy integration), they are not necessarily representative of industrial scale production, in contrast to data available for fossil-based polyols considered as a reference. Therefore, a fully appropriate and consistent comparison with the other scenarios assessed for this case study could not be ensured. A partly bio-based alternative to fossil-based insulation material was assessed in Scenario 6, where soy-based PUR was considered. This is obtained by replacing fossil-based polyols with soy-based polyols in the PUR production process. Owing to the fact that the market/production of soy-based polyols is still relatively small, little information was available to compile a LCI of the production process. Therefore, the results obtained may not be representative of full-scale industrial production and should be used carefully.



**Table 9.2.** LCA scenarios assessed for the insulation material screening case study. Note that CO<sub>2</sub>-PUR and Bio-PUR are only partly CO<sub>2</sub>- and bio-based. R1: recycled content.

Scenario	Polymer	Monomer(s)	Feedstock	End of Life options <sup>(1)</sup>
1 - Conventional polymer 1	PUR	PO <sup>(2)</sup> EO <sup>(3)</sup> MDI <sup>(4)</sup>	Fossil-based (oil/natural gas)	Recycling Incineration Landfilling
2 - Conventional polymer 2	EPS	Styrene	Fossil-based (oil/natural gas) Waste EPS (R1=2%)	Recycling Incineration Landfilling
3 - Alternative polymer 1	R-EPS (100% recycled content)	Styrene	Waste EPS (post-consumer)	Recycling Incineration Landfilling
4 - Alternative polymer 2	R-PET (100% recycled content)	MEG <sup>(5)</sup> PTA <sup>(6)</sup>	Waste PET (post-consumer)	Recycling Incineration Landfilling
5 - Alternative polymer 3	CO <sub>2</sub> -PUR	PO <sup>(2)</sup> EO <sup>(3)</sup> MDI <sup>(4)</sup>	Fossil-based CO <sub>2</sub> Fossil-based (oil/natural gas)	Recycling Incineration Landfilling
6 - Alternative polymer 4	Bio-PUR	PO <sup>(2)</sup> EO <sup>(3)</sup> MDI <sup>(4)</sup>	Soybean (EU) Fossil-based (oil/natural gas)	Recycling Incineration Landfilling

(1) The impacts of the scenarios were individually assessed for each End of Life option listed, as well as for a combination of such options reflecting as far as possible the average situation at the EU level (i.e., 45% incineration and 55% landfilling conforming with the assumptions taken in the pilot PEFCR pilot study on insulation materials, see Avnir, 2019)

(2) PO: propylene oxide

(3) EO: ethylene oxide

(4) MDI: methylene diphenyl diisocyanate

(5) MEG: mono ethylene glycol

(6) PTA: purified terephthalic acid

## 9.2 Functional Unit and reference flow

The main function of the studied article is providing insulation in buildings pitched roofs (Table 9.2). The functional unit of this case study was thus defined as “delivering an insulation board with area equal to 1 m<sup>2</sup> that provides a thermal resistance (R) equal to 7.14 m<sup>2</sup>·K·W<sup>-1</sup> (or 0.14 W m<sup>-2</sup>K<sup>-1</sup>) with a designed life span of 50 years”. This functional unit is typically used in LCAs of insulation solutions and provides information on 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, focusing on the insulating and environmental properties of the material under investigation (Avnir 2019; Pargana et al., 2014). The thermal resistance value of 7.14 m<sup>2</sup>·K·W<sup>-1</sup> (or 0.14 W m<sup>-2</sup> K<sup>-1</sup>) reflects the performance required under temperate EU conditions, typically ranging from 3 to 10 m<sup>2</sup>·K·W<sup>-1</sup> (or 0.1 to 0.33 W m<sup>-2</sup> K<sup>-1</sup>); for additional explanations, the reader is referred to Avnir (2019). The reference flow of each scenario, i.e. the amount of material required in order to fulfil the functional unit, was calculated departing from the

given service and the specific properties of the materials (Reference flow= $R \cdot \lambda \cdot \rho \cdot A$ ). Table 9. summarises the reference flow in each scenario. Data for material properties were retrieved from actual producers.

**Table 9.2.** Definition of the functional unit for insulation boards LCA scenarios.

Aspect	Description
“What” (function(s) or service(s) provided)	Providing insulation in buildings pitched roofs.
“How much” (extent of the function(s) or service(s))	Delivering an insulation board with area equal to 1 m <sup>2</sup> that provides a thermal resistance (R) equal to 7.14 m <sup>2</sup> ·K·W <sup>-1</sup> (or 0.14 W m <sup>-2</sup> K <sup>-1</sup> ).
“How well” (expected level of quality)	Providing the given thermal resistance in buildings pitched roofs during the building life time.
“How long” (duration/lifetime of the function or service)	Throughout the insulation life span, here assumed to be 50 years for buildings.
“Where” (location/geography of the service)	The thermal resistance value of 7.14 m <sup>2</sup> ·K·W <sup>-1</sup> (or 0.14 W m <sup>-2</sup> K <sup>-1</sup> ) reflects the performance required under temperate EU conditions, typically ranging from 3 to 10 m <sup>2</sup> ·K·W <sup>-1</sup> (or 0.1 to 0.33 W m <sup>-2</sup> K <sup>-1</sup> ).

**Table 9.3.** Reference flow for insulation boards LCA scenarios.

Polymer	Thermal conductivity ( $\lambda$ ) (W·m <sup>-1</sup> ·K <sup>-1</sup> )	Density ( $\rho$ ) (kg·m <sup>-3</sup> )	Area (A) (m <sup>2</sup> )	Reference flow ( $R \cdot \lambda \cdot \rho \cdot A$ ) <sup>(1)</sup> (kg)
PUR <sup>(2)</sup> (all types of feedstock)	0.022	31.14	1	4.9
EPS <sup>(3)</sup> (all types of feedstock)	0.031	15	1	3.3
R-PET (R1=100%) <sup>(4)</sup>	0.028	48	1	9.6

(1)  $R = 7.14 \text{ m}^2 \cdot \text{K} \cdot \text{W}^{-1}$

(2) From PEFCR pilot study on insulation, Avnir (2019).

(3) Based on EUMEPS (2020) and PEFCR study on insulation, Avnir (2019).

(4) Based on technical data from company’s website; Armaform (2020).

### 9.3 System boundary

In all scenarios, the system boundary was set in order to cover the most relevant stages and processes of the full product life cycle (cradle-to-grave perspective), as illustrated in Figures 9.1 to 9.6:

- *Feedstock Supply*<sup>148</sup> – covering extraction, transport and possible refining of crude oil and natural gas (fossil-based polymers), crop cultivation (bio-based polymers), collection, transport and sorting of plastic waste (recycled polymers), CO<sub>2</sub> capture, as well as transport of these feedstock sources to downstream conversion or

<sup>148</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

1 utilisation processes (e.g. naphtha cracking, polymer recycling, soybean refining,  
2 polyol production);

- 3 • *Polymer Production*<sup>149</sup> – covering all the activities associated with the conversion  
4 of feedstock materials into the relevant monomer(s) and final polymer, including  
5 any transport among these activities and, where relevant, final transport of  
6 polymer granulate to downstream manufacturing processes;

- 7 • *Article Production*<sup>150</sup> – including insulation board manufacturing through suitable  
8 conversion processes, i.e. formation, cutting and packing of the boards;

- 9 • *Distribution* – including transport of insulation boards from the manufacturing site  
10 to the retailers and from these to the final consumers;

- 11 • *End of Life* – covering collection, transport, recycling, incineration, or disposal of  
12 insulation boards after use, including any avoided processes from virgin material  
13 or energy substitution.

14 A different nomenclature was applied for some life cycle stages compared to the default  
15 nomenclature specified in the methodological document (and in the PEF method) to make  
16 it more relevant for the investigated supply chains and the project scope. Moreover, the  
17 default “Raw Material Acquisition and Pre-processing” stage was further split into two  
18 separate sub-stages (i.e. Feedstock Supply and Polymer Production), to allow  
19 disaggregating the impacts of feedstock supply from downstream conversion processes,  
20 and hence to better appreciate any differences among the use of different feedstock  
21 sources (in line with the project goal).

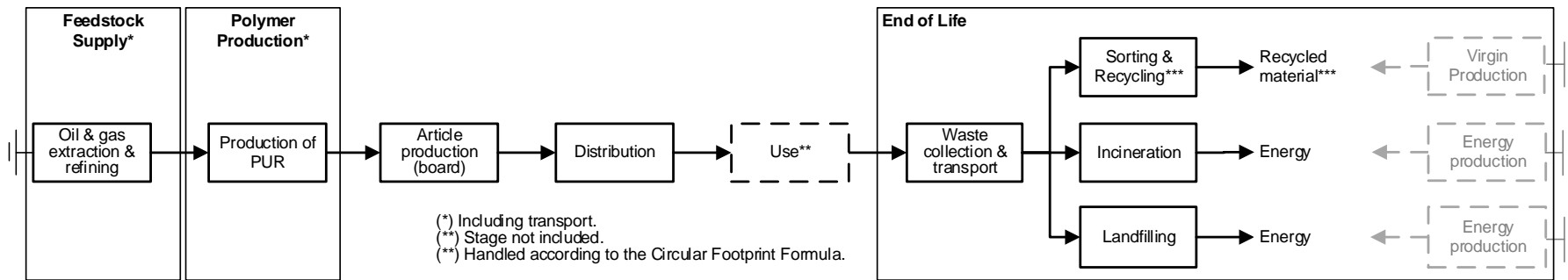
22 Relevant transport activities between the different life cycle stages were considered in  
23 the study, as was the indirect land use change following demand for cropping land (only  
24 for additional environmental information, not as part of main model/results). Note that  
25 the use stage has been omitted as it can be reasonably assumed that the use phase for  
26 these building insulation materials incurs negligible impacts. This was also the case in the  
27 PEFCR pilot study on insulation (Avnir, 2019).

28 Finally, it has to be noted that additives were not included in the assessment, due to the  
29 lack of complete and consistent data on the use of additives in the production of  
30 insulation boards, of the examined polymers and of plastics in general, as well as on their  
31 release and fate over the product life. This is acknowledged as a limitation of this study,  
32 as additive production can account for a non-negligible portion of cradle-to-gate Climate  
33 Change impact and energy demand, which is up to 46% for (starch-based) polymer  
34 grades including larger shares of additives (Broeren et al., 2017). Moreover, additives  
35 can also be relevant at the End of Life stage, where they can be released, as such or  
36 after degradation/conversion into different compound(s), in the environment (e.g. the  
37 soil in case of biodegradable plastics routed to biological treatments or subject to in-situ  
38 degradation).

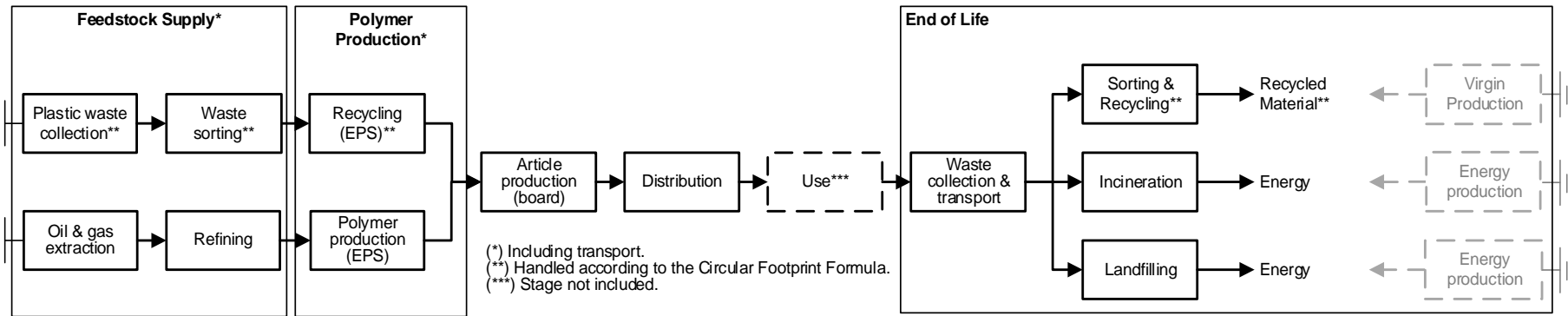
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<sup>149</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

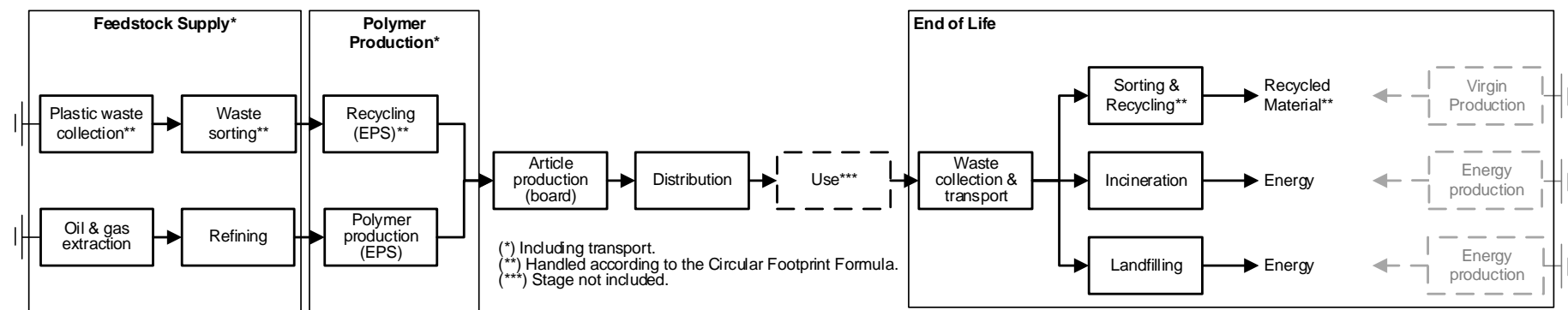
<sup>150</sup> Corresponding to the default stage of “Manufacturing” specified in the Method and in the PEF framework.



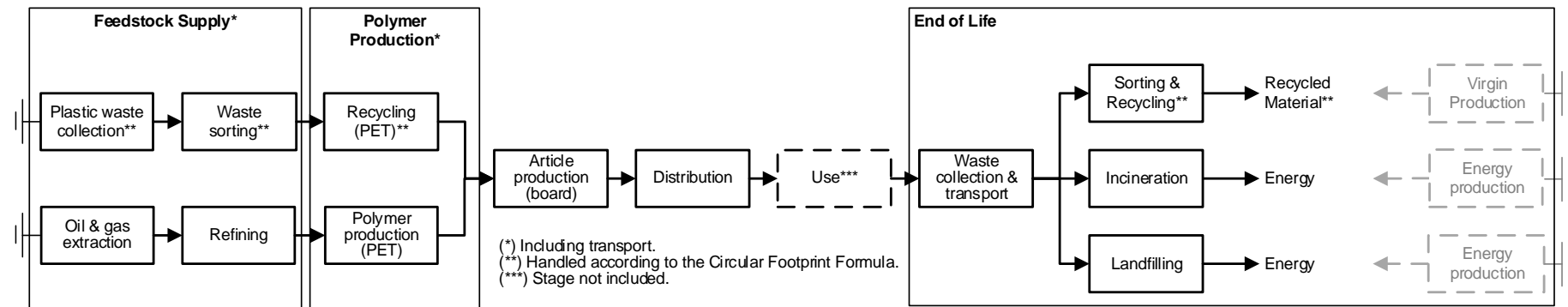
**Figure 9.1.** System boundary for fossil-based PUR insulation boards (Scenario 1). The current average EU EoL for insulation materials consists of landfilling (55%) and incineration (45%) in agreement with PEFCR pilot on insulation materials (Avnir, 2019).



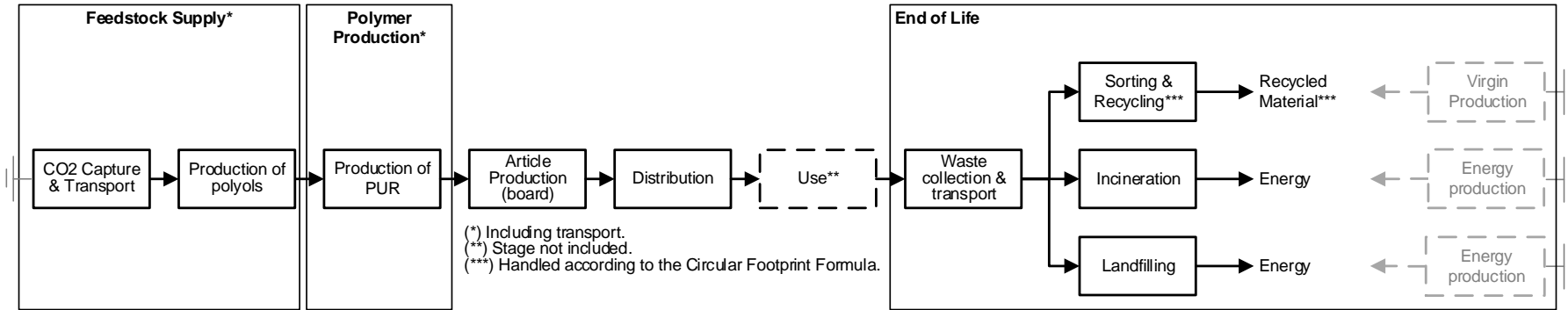
**Figure 9.2.** System boundary for fossil-based EPS insulation boards (Scenario 2). The current average EU EoL for insulation materials consists of landfilling (55%) and incineration (45%) in agreement with PEFCR pilot on insulation materials (Avnir, 2019).



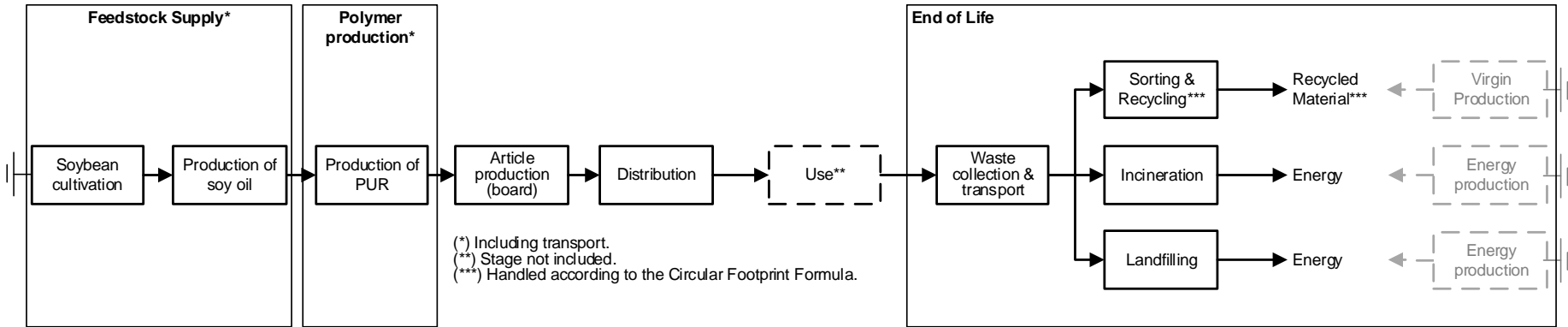
**Figure 9.3.** System boundary for fossil-based R-EPS insulation boards (Scenario 3). The current average EU EoL for insulation materials consists of landfilling (55%) and incineration (45%) in agreement with PEFCR pilot on insulation materials (Avnir, 2019).



**Figure 9.4.** System boundary for fossil-based R-PET insulation boards (Scenario 4). The current average EU EoL for insulation materials consists of landfilling (55%) and incineration (45%) in agreement with PEFCR pilot on insulation materials (Avnir, 2019).



**Figure 9.5.** System boundary for fossil CO<sub>2</sub>-based PUR insulation boards (Scenario 5). The current average EU EoL for insulation materials consists of landfilling (55%) and incineration (45%) in agreement with PEFCR pilot on insulation materials (Avnir, 2019).



**Figure 9.6.** System boundary for bio-based bio-PUR insulation boards (Scenario 6). The current average EU EoL for insulation materials consists of landfilling (55%) and incineration (45%) in agreement with PEFCR pilot on insulation materials (Avnir, 2019).

## 9.4 Life Cycle Inventory

This section describes the overall approach for developing the Life Cycle Inventory of the analysed scenarios, along with the related assumptions and data sources. The description is separated by major lifecycle stages, in the following sub-sections (9.4.1 – 9.4.6). A detailed list of processes, related data sources, and other modelling details is provided in Tables B.6.1 to B.6.6 in Annex B.6.

### 9.4.1 Feedstock Supply Stage

#### 9.4.1.1 Fossil-based polymers

For conventional fossil-based polymers (PUR, EPS, PET), the stage of Feedstock Supply includes the processes of crude-oil and natural gas extraction and transport, naphtha production in crude oil in refineries, and its subsequent transport to downstream conversion processes. For the detailed description of the modelling of these activities and of the datasets applied for this purpose, the reader is referred to Section 4.4.1.1.

#### 9.4.1.2 Recycled polymers

For recycled polymers (R-EPS and R-PET), Feedstock Supply consists of collection of post-consumer plastic waste of the relevant polymer, and subsequent transport and sorting in specific facilities. These processes were modelled as described in Section 9.4.5 (addressing End of Life modelling), and refer to collection and transport of separately collected plastic waste at the municipal level (in the absence of specific data for plastic waste collection from the different sectors it may come from), as well as to sorting of mixed plastic waste in dedicated facilities. These processes were implemented in the lifecycle model according to the Circular Footprint Formula (CFF), which is the default approach to handle recycling situations in the Product Environmental Footprint (PEF) context. According to this approach, only 50% of the burdens from collection, transport, sorting and recycling were assigned to the recycled material input used in the system (i.e. factor  $A = 0.5$ ), the rest being assigned to the system providing material for recycling.

#### 9.4.1.3 Bio-based polymers

For bio-based PUR, the stage of Feedstock Supply (soybean oil) includes cultivation of soybean and subsequent transport to further processing in the same country. The EF compliant dataset *Crude soybean oil; from crushing (pressing and solvent extraction), production mix; at plant; EU+28* is used. This describes the production of crude soybean oil, soybean hulls and soybean meal from a soybean crushing process, in EU-28 + EFTA. Considered activities include inputs of soybeans and associated transport, electricity, water, heat from natural gas and auxiliary materials such as hexane and outputs of waste and wastewater to treatment. Capital goods (for cultivation and transport) as well as their End of Life are included. The crushing of soybeans normally occurs in a series of process steps, including dehulling of the soybeans, mechanical crushing of the beans with partial extraction of the oil, and further extraction of the oil using hexane as a solvent with subsequent hexane recovery steps.

#### 9.4.1.4 CO<sub>2</sub>-based polymers

Activities related to Feedstock Supply for CO<sub>2</sub>-based PUR production include capture, purification and liquefaction of CO<sub>2</sub> from flue gas of coal-based power plants (or the sole purification and liquefaction of the almost pure gaseous CO<sub>2</sub> output from Ammonia production) and its transport to downstream conversion (utilisation) facilities in a liquefied form via pipelines (assumed to be the most suitable transport form for industrial use at large scale).

As a base case, CO<sub>2</sub> was assumed to be a waste from a CO<sub>2</sub> source (coal-fired power plants or ammonia production plants), according to the discussion reported in Section 4.4.4 of the Method (Report I), and especially considering that raw gaseous CO<sub>2</sub> has no economic value at the point of its arising. This assumption is also in line with the discussion by Giegrich et al. (2018), who argue that CO<sub>2</sub> can be considered as a waste if it has no value before capturing, or if its supply is much larger than its demand in the context of the CO<sub>2</sub> emitter (as it is currently the case for power plants and many other CO<sub>2</sub> sources). In this perspective, the subsequent processes of capture<sup>151</sup>, purification, compression, transport and utilisation constitute the components of a recycling chain aimed at converting waste CO<sub>2</sub> into a useful, CO<sub>2</sub>-based product (i.e. Polyols and, later, Polyurethane), ultimately replacing an equivalent product from primary resources. Therefore, the Circular Footprint Formula was applied, as in any other recycling situation, to model the environmental burdens of the feedstock CO<sub>2</sub> (as better discussed in Section 4.4.4 of the Method (Report I)). Alternative approaches were explored as a sensitivity analysis (Section 9.7.7.2), applying also a wider system perspective where the CO<sub>2</sub>-based product (i.e. CO<sub>2</sub>-based Polyols) is considered, along with the main product of the CO<sub>2</sub> source (i.e. coal-based electricity) a co-product of the overall Carbon Capture and Utilisation (CCU) system.

The application of the CFF implied that a share of the burdens of the capture and utilisation (i.e. “recycling”) process-chain (including capture, purification, liquefaction, transport and utilisation) were assigned to the CO<sub>2</sub>-based product. Conversely, no burdens from any upstream activity related to the supply and combustion of the CO<sub>2</sub>-providing fossil-based feedstock (e.g. coal extraction and combustion) were attributed to the product, as these activities take place before waste CO<sub>2</sub> arising. On the other hand, we assumed that the CO<sub>2</sub>-based product carries a share of the burdens associated with the primary production of the conventional product it replaces, including supply of the respective primary feedstock. In principle, the most proximate point of substitution within the supply chain should be considered, which in this case could be identified at the level of Propylene Oxide (with captured CO<sub>2</sub> partially replacing Propylene Oxide in Polyol production). However, since the subsequent CO<sub>2</sub>-based Polyol synthesis partially differ from the conventional synthesis route in terms of material and energy inputs and outputs (and different data were indeed applied to the modelling of these routes in this study – see Sections 9.4.2.1 and 9.4.2.4), it was considered more appropriate to assume substitution at the Polyol level (with CO<sub>2</sub>-based Polyols replacing conventional, fossil-based Polyols). The share of burdens of the two production pathways assigned to the CO<sub>2</sub>-based product depends on the value of the A factor, which should be defined by taking into account the relation between supply and demand of the “recycled” (in this case, CO<sub>2</sub>-based) product. Since there is currently no established market for CO<sub>2</sub>-based Polyols, the evaluation was conducted assuming a “neutral” situation of equilibrium between supply and demand of this CO<sub>2</sub>-based product (i.e. A=0.5). Considering the current large availability of CO<sub>2</sub> sources, such a situation can be reasonably expected to occur once an established demand for CO<sub>2</sub>-based Polyols would exist. According to this choice, only 50% of the burdens from the processes of capture, purification, liquefaction and transport of waste CO<sub>2</sub> were assigned to the CO<sub>2</sub>-based product (i.e. Polyols). On the other hand, the latter carries 50% of the burdens associated with the supply of fossil-based feedstock required for the production of fossil-based polyols (modelled as described in Section 9.4.1.1).

The CO<sub>2</sub> capture, purification and liquefaction processes were modelled based on the *ecoinvent* dataset “[RER] Carbon dioxide, liquid”, which refers to CO<sub>2</sub> extraction from (waste) gas streams of industrial production processes by means of a 15-20% Monoethanolamide (MEA) solution, followed by purification and liquefaction. Compared to the original dataset, the electricity consumption was updated with a more representative

<sup>151</sup> Only if capture is purposefully carried out at the CO<sub>2</sub> source to enable its downstream utilisation. This is not the case, for instance, of Ammonia production plants, where CO<sub>2</sub> separation/extraction from the gaseous mixture of CO<sub>2</sub> and Hydrogen generated in the process is necessarily carried out as an integral part of the process itself to allow subsequent use of Hydrogen for Ammonia synthesis.



value (1.32 MJ/kg CO<sub>2</sub>), calculated as the average of values reported in Von der Assen et al. (2015) for CO<sub>2</sub> extraction from flue gases of coal-based power plants using MEA as a solvent. On the other hand, the default values applied in the dataset for capture efficiency, MEA consumption, water flows, and air emissions were considered representative also for extraction from flue gas of power plants (except for Methane emissions that were removed). Finally, background datasets related to energy supply (in this case only electricity) were replaced with EF datasets, while infrastructure processes (related to chemical factory construction) were removed, to improve reliability of LCIA results in the Ozone Depletion impact category.

Transport of liquefied CO<sub>2</sub> to downstream industrial users via pipelines was assumed to take place along a distance of 300 km, in line with the “CO<sub>2</sub> deserts map” reported in Von der Assen et al. (2015) for CO<sub>2</sub> sources located in Europe. According to the latter, considering only CO<sub>2</sub> sources available today, the current European demand of 50 Mt CO<sub>2</sub>/year can be met by CO<sub>2</sub> sources located at a distance not larger than 300 km from hypothetical users in central Europe (and demanding up to 5 Mt CO<sub>2</sub>/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”. 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 datasets for electricity generation in the EU. Moreover, leakage of natural gas and the related air emissions were replaced with leakage and emissions of CO<sub>2</sub>.

## 9.4.2 Polymer Production Stage

The Polymer Production stage covers the activities of feedstock processing into relevant intermediates and monomer(s), the polymerisation process, as well as any transport among these activities and final transport of polymer granulate to the insulation boards manufacturing site. The following subsections (9.4.2.1 – 9.4.2.5) describe how these activities have been modelled in the present case study.

### 9.4.2.1 Fossil-based polymers

Polyurethane (PUR) 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. Production of 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.

For conventional production of (fossil-based) rigid polyether polyols and MDI, EF-compliant LCI datasets were used to model supply at the EU-28 level. For the final production of (fossil-based) polyurethane rigid foam boards departing from such fossil-based precursors, disaggregated EF-compliant datasets were not available. A foreground dataset was then developed based on the ecoinvent dataset *RER polyurethane, rigid foam, at plant* (per 1 kg PUR: 0.62 kg MDI, 0.39 kg polyols, 0.42 kWh electricity, 0.054 kg pentane) using EF-compliant background datasets to represent inputs (chemicals, energy, etc.) and outputs to/from the process. It should be noticed that this is exactly the same as the LCI dataset reported for PUR rigid foam by PlasticsEurope (2004).

Polystyrene production was modelled using a disaggregated TS-datasets (i.e. disaggregated for feedstock supply and subsequent polymer production) for the production of polystyrene ([DE] *Polystyrene granulate (PS), polymerisation of styrene | production mix, at plant* | 1.05 g/cm<sup>3</sup>, 104.15 g/mol per repeating unit {077f6dea-b740-4604-afec-84120c4655ab}). The expansion process of PS to EPS was modelled according with the ecoinvent process *GLO polystyrene foam slab for perimeter insulation* (per 1 kg

EPS: 1 kg PS, 0.56 kWh electricity, 5.32 MJ heat, 0.02 kg polyethylene film and transport) using EF-compliant background datasets to represent inputs (chemicals, energy, etc.) and outputs to/from the process.

Polyethylene production was modelled using the EF-compliant dataset for the production of PET namely *Polyethylene terephthalate bottle grade granulate (PET) via PTA*. While an EF-dataset for amorphous PET was available, this was not used because of unreliable results obtained for a few impact categories (e.g. water scarcity and mineral resource).

#### 9.4.2.2 Recycled polymers

For recycled EPS production (R-EPS), a mechanical recycling of EPS was modelled on the basis of the information available in PWC (2011). Accordingly, shredding, densification and palletisation of EPS-waste to produce densified PS pellets (for further utilisation as feedstock in EPS production) overall requires 270 kWh/t waste treated. Slightly lower energy consumptions could be calculated from available catalogues of plastic recycling machineries and equipment (shredders, densifier and pelletisers for EPS and similar low-density plastic materials; ca. 100-150 kWh/t waste treated depending upon machineries capacity; see technical specifications available in Promeco 2019). EF-compliant datasets were used to represent energy consumption. Similarly to PET recycling, we assumed an efficiency of the recycling process equal to 85%; the residues were assumed to be disposed of according to the same End of Life ratio assumed in the rest of the modelling (55% landfilling and 45% incineration). Capital goods were excluded due to lack of data.

For recycled PET production (R-PET), an EF-compliant dataset was available (*Polyethylene terephthalate (PET) granulate secondary; no metal fraction*), representing the burdens of the mechanical recycling process of post-consumer plastic waste via grinding, washing, metal separation and pelletizing, with an overall process efficiency of 85.5%. Contrarily to the case study on bottles, PET granules from mechanical recycling were here assumed to be directly used for manufacturing of insulation boards without requirement for further upgrading.

Following the approach adopted in the PEF context to model plastic recycling situations (Circular Footprint Formula, CFF) and in the absence of reliable figures for PET/EPS insulation plastics, it was assumed that the recycled material used in input carried only 50% of the burdens of the recycling process (this means that the factor  $A = 0.5$ ) similarly to the assumptions taken in the earlier described case studies. Likewise, such recycled material carried a share ( $1-A = 0.5$ ) of the production burdens associated with the replaced virgin material (i.e. the same burdens that would have been credited to the previous life cycle providing the recycled material). The  $Q_{sin}/Q_p$  factor was assumed equal to 1, i.e. the allocated share of virgin production impacts was equal to 50% ( $A \times Q_{sin}/Q_p = 0.5 \times 1 = 0.5$ ).

#### 9.4.2.3 Bio-based polymers

The production of partly bio-based PUR (bio-PUR) was modelled by substituting the precursor (fossil-based) polyol with soy-based polyol. The foreground LCI representing the production process of soy-based polyols was developed on the basis of the information detailed in Omnitech International (2010). Accordingly, the production of 1 kg of refined soybean oil requires 1.042 kg crude soybean oil, 0.0023 kg NaOH, 0.156 kg water, 0.036 MJ electricity and 0.133 MJ heat. The production of 1 kg polyol requires 1 kg refined soybean oil, 1.5 MJ electricity and 2.24 MJ heat from natural gas. EF-compliant background datasets were used to model inputs (chemicals, energy, etc.) and outputs to/from the process. Within the soy oil refining process, no relevant co-products were generated (negligible amounts of soap). Soy-based polyols were then used as input to the PUR production process in place of fossil-based polyols, as described earlier (Section 9.4.2.1). In this, perfect substitutability of fossil-based polyols with soy-based polyols was assumed, in the absence of more specific information.

#### 9.4.2.4 CO<sub>2</sub>-based polymers

The modelling of partly CO<sub>2</sub>-based PUR production (CO<sub>2</sub>-PUR) was based on the process described in Fernandez-Dacosta et al. (2017). In this, Propylene Oxide is partly replaced by CO<sub>2</sub> (circa 16.5%). A dataset was therefore developed departing from the information reported in Fernández-Dacosta et al. (2017), using EF-compliant background datasets whenever possible. The production of 1 kg Polyol requires 1.6E-1 kg CO<sub>2</sub>, 8.08E-1 kg Propylene Oxide, 2.16E-2 kg Glycerine, 6.19E-2 kg Propylene Glycol, 5.98E-2 MJ steam, 1.14E-4 Double Metal Cyanide (DMC) catalyst. Mineral material (Zeolite) is generated as waste (1.14E-4 kg); for more details see Supplementary Information of Fernández-Dacosta et al. (2017), Table S.15. The Polyols were then used as input to the PUR production process in place of fossil-based Polyols, as described earlier (Section 9.4.2.1). In this, perfect substitutability of fossil-based Polyols with CO<sub>2</sub>-derived Polyols was assumed, in the absence of more specific information.

#### 9.4.2.5 Transport of polymer granulate to the article production site

Modelling of transport of polymer granulate from the polymerisation plant to the insulation board manufacturing site in Europe is based on the default transport scenario (distances and vehicle types) specified in the EF context and in the Method (Report I) for the route supplier-to-factory. All polymers considered in this case study are produced in EU (virgin EPS and PUR import shares to EU are about 5% and 2%, respectively), thus the following average routes were considered:

- (i) 130 km by lorry (total weight >32 t; Euro 4);
- (ii) 240 km by train (average freight); and
- (iii) 270 km by ship (barge).

#### 9.4.3 Article Production Stage

For EPS-based insulation boards, the stage of article production involves the operations of expansion of PS into EPS, cutting and final packaging of the boards. This was modelled using the EI database *polystyrene foam slab for perimeter insulation* (5.32 MJ heat, 0.02 kg PE film and 2.02 MJ electricity per kg of EPS board) using EF-compliant background datasets to model the inputs (materials, energy, etc.). Following expansion operations, air emissions of pentane and CO<sub>2</sub> occur (0.00107 and 0.078 kg, respectively). For PUR- and PET-based insulation boards, the stage of article production involves the operations of cutting and final packaging of the boards. This was modelled as for EPS, assuming that similar operations apply (0.02 kg PE film and 2.02 MJ electricity per kg of board). The production of R-PET based insulation boards was based on the technology described in Ingrao et al. (2019) representing a full-scale production facility located in Italy. The process involves production of recycled PET fibres from recycled PET flakes (2.53 MJ heat for drying and 2.14 kWh electricity for extrusion to produce 1 kg R-PET fibre) and the following utilisation of fibres to produce boards (1.025 kg R-PET fibres, 1.94 MJ heat for drying, 0.4 kWh electricity and 0.03 kg PE film to produce 1 kg R-PET board). Capital goods were not included due to lack of data. For additional details on the technology and process, please refer to the original publication from Ingrao et al. (2019).

#### 9.4.4 Distribution Stage

The distribution of the article from the manufacturing site to the final user was modelled based on the default transport scenario specified in the PEF context (and in the Method) for the route *factory → retail → final client*. A volume-limited transport was assumed for all. The following routes were thus considered:

- i) 1200 km by lorry from factory to retailers assuming volume-limited transport;
  - ii) 10 km by van for 100% of the roundtrips from retailers to final users.
- Contrarily to the case studies on small products (e.g. bottles), it is thus

assumed that all the local transport retail-to-final user occurs using  
motorised vehicles (van).

In the absence of a specific dataset for volume-limited transport via truck >32t, the  
dataset for volume-limited transport with van <7.5t was used instead, conforming to the  
methodology described in Report I.

#### **9.4.5 End of Life Stage**

This section describes the modelling of the End of Life stage of insulation boards. In  
particular, Section 9.4.5.1 focuses on the definition of the EU-average End of Life  
scenario, which is considered as a base case for the calculation of the potential impacts  
of the compared LCA scenarios. The remaining sections (9.4.5.2 – 9.4.5.5) address the  
modelling of waste collection and transport, and of the different End of Life options  
explored in the study. Finally, Section 9.4.5.6 provides case-specific details on the  
estimate of the potential contribution of the insulation boards to macro-plastics formation  
at the End of Life (including product litter) and micro-plastic generation throughout the  
supply chain.

##### **9.4.5.1 End of Life scenario**

The same EU-average End of Life scenario was considered for all the investigated  
insulation materials, regardless of the type of feedstock used for polymer production  
(fossil resources, plastic waste or biomass), as it can be reasonably assumed that this  
does not affect the viable End of Life options for the insulation boards made of this  
material, nor those currently applied to them. Specifically for insulation materials, very  
limited data on the current recycling rates and techniques, when applied, exist. For PUR  
materials, including all applications (not only insulation), ISOPA (2005) reports a  
recycling rate in the order of 9.5% for year 2004. For EPS waste, including all market  
applications (not only insulation), EUMEPS (2018) reports a recycling rate in the order of  
8.5%. After careful consideration of the sources, these figures were nevertheless  
considered obsolete and/or not supported by robust background data and analyses.  
Further, no specific rates for building insulation waste were provided. Having this in  
mind, the EoL for all the investigated insulation materials was here assumed to consist of  
landfilling (55% of the collected waste) and incineration (45% of the collected waste),  
conforming with the assumptions of the PEFCR pilot study on the current End of Life of  
insulation products (Avnir 2019). Yet, we also illustrate the results for the case of 100%  
landfilling, 100% recycling, and 100% incineration in Section 9.5.

##### **9.4.5.2 Modelling of waste collection and transport**

Collection, transport, and sorting were modelled similarly to the post-consumer plastic  
products as described in section 4.4.5.2. It is acknowledged that collection and sorting of  
insulation materials may differ due to, e.g. selective demolition practices and/or *ad hoc*  
management schemes for construction and demolition waste of which insulation  
materials are part. However, related technologies and processes are currently no widely  
established and mostly under research and development. Owing to this, little or no  
information, the authors are aware of, is currently available with respect to processes  
and related life cycle inventories.

##### **9.4.5.3 Modelling of sorting and recycling**

As earlier mentioned, sorting was modelled as described in section 4.4.5.3 assuming that  
the insulation materials, once collected, require to undergo further sorting to obtain a  
number of individual plastic-polymer streams.

Mechanical recycling of PUR was modelled on the basis of the information 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 scrap particles  
(shredded; size ~1 cm) with di-isocyanate MDI followed by form-shaping at 100-200°C,

30-200 bar. PU construction boards with excellent water and moisture resistance are obtained, or insulation panels for use in new refrigerators or freezers. This is also illustrated as one of the likely pathways in ISOPA (2005). Building upon the information reported in Zevenhoven (2004), the recycling process requires shredding (21 kWh/t waste treated; similarly to EPS treatment), pressure bonding (250 kWh/t waste treated; calculated assuming to achieve a mechanical pressure of 200 bar on a volume of 0.032 m<sup>3</sup>/kg PUR assuming a PUR-board density of 31.14 kg/m<sup>3</sup> and a conversion factor of 0.7 between electricity and obtainable mechanical energy) and addition of diphenyl methane diisocyanate as a binder (MDI; 0.1 kg/kg recycled PUR, i.e. ca. 10% of the recycled material). Likewise PET recycling, we assumed an efficiency of the recycling process equal to 85%; the residues were assumed to be disposed of according to the same EoL ratio assumed in the rest of the modelling (55% landfilling and 45% incineration).

Mechanical recycling of EPS was modelled on the basis of the information available in PWC (2011). Accordingly, shredding, densification and pelletization of EPS-waste to produce densified PS pellets (for further utilisation as feedstock in EPS production) overall requires 270 kWh/t waste treated. Slightly lower energy consumptions could be calculated from available catalogues of plastic recycling machineries and equipment (shredders, densifier and pelletisers for EPS and similar low-density plastic materials; ca. 100-150 kWh/t waste treated depending upon machineries capacity; see technical specifications available in Promeco 2019). EF-compliant datasets were used to represent energy consumption. Similarly to PET recycling, we assumed an efficiency of the recycling process equal to 85%; the residues were assumed to be disposed of according to the same EoL ratio assumed in the rest of the modelling (55% landfilling and 45% incineration). Capital goods were excluded due to lack of data.

For mechanical recycling of PET, an EF-compliant dataset was available (*Polyethylene terephthalate (PET) granulate secondary; no metal fraction*), representing the burdens of the mechanical recycling process of post-consumer plastic waste via grinding, washing, metal separation and pelletizing, with an overall process efficiency of 85.5%. Contrarily to the case study on bottles, PET granules from mechanical recycling were here assumed to be directly used for manufacturing of insulation boards without requirement for further upgrading (no polycondensation).

Following the approach adopted in the PEF context to model plastic recycling situations (Circular Footprint Formula), the recycled material output was assumed to replace the corresponding virgin polymer, whose primary production burdens were credited to the system (see system boundary diagrams; Figure 9.1 to 9.6). A substitution ratio equal to 1 ( $Q_{Sout}/Q_p$ ) was considered in the absence of more specific indications. Only 50% of the burdens of the EoL sorting and recycling processes were assigned to the system ( $A = 0.5$ ). Similarly, only 50% of the benefits from avoided virgin material production were assigned to the system itself.

#### **9.4.5.4 Modelling of incineration**

Incineration was modelled on the basis of a common, aggregated, EF compliant dataset representing incineration of plastic waste in an average EU MSW incineration namely *[EU28] Waste incineration of plastics (unspecified) fraction in municipal solid waste (MSW)*. For a detailed description of the modelling principles for incineration in Gabi and of the downstream substitution of electricity and heat through energy recovery from the waste, the reader is referred to section 4.4.5.4. It should be noticed that the recovery of energy from waste was correlated to the specific LHV of the waste material in input.

#### **9.4.5.5 Modelling of landfilling**

Landfilling was modelled on the basis of a common, aggregated, EF compliant dataset representing disposal of plastic waste in a managed, municipal waste landfill (*[EU-28+EFTA] Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific sites)*). For a detailed description of the process, the reader is referred to section 4.4.5.5.

#### 9.4.5.6 Contribution to littering and micro-plastic generation

The contribution to macro- and micro-plastics loss and release of the analysed building insulation boards scenarios was estimated according to the framework outlined in Annex B of the Method (Report I), and summarised in Section 4.4.5.6 of this report in terms of underlying quantification methods and approaches. In this section, the case-specific parameters considered to apply the latter to the assessed building insulation boards LCA scenarios are reported.

##### 9.4.5.6.1 PLP method

Regarding the estimate of the loss and release of macro-plastics at the End of Life stage (from product littering and waste mismanagement), the product-specific parameters reported in Table 9.4 were considered to apply the first level of the framework ("*PLP method*") to building insulation boards (derived from Peano et al., 2020). Note that for the scenario representing recycled PET release rates are lower due to a higher economic value of the material.

**Table 9.4.** Case-specific parameters of the PLP method applied to the building insulation boards LCA scenarios.

Parameter <sup>(1)</sup>	Value	R-PET
Littering rate ( $LR_{lit}$ ) (%)	0	0
Release rate to ocean ( $Rel_{ocean}$ ) (%)	5	1
Release rate to the terrestrial environment ( $Rel_{terenv}$ ) (%)	95	1

<sup>(1)</sup> For details on the meaning of each parameter, refer to Annex B of the Method (Report I).

No case-specific parameters are required to calculate the supply-chain loss and release of micro-plastics to ocean and to the terrestrial environment by applying the *PLP method*. Indeed, the contribution of the considered micro-plastics sources (pellet loss and tire abrasion) depends on the magnitude of the relevant life cycle processes (i.e. polymer production and road transport) in the specific building insulation boards scenario (which in turns depend on the reference flow<sup>152</sup> of each scenario). However, the quantification is made by means of default parameters that are not affected by the type of product, polymer or feedstock source (as reported in Annex B of the Method (Report I)).

##### 9.4.5.6.2 Expanded PLP method

The parameters required to calculate the micro-plastics loss and release from marine coatings and road markings are not case-specific, and are reported in Annex B of the Method (Report I). Similar to the other considered micro-plastics sources, the contribution of these additional sources depends on the intensity of relevant (transport) processes in the specific product inventory, by means of the above-mentioned case-unspecific parameters.

As for the calculation of the additional contribution of plastic waste exported for recycling to the loss and release of macro-plastics from waste mismanagement, this case study considers that there is no recycling stream at the End of Life of the product. Thereby, no additional calculation due to exported plastic waste for recycling has been performed.

#### 9.4.6 Calculation of iLUC impacts

As a base case, the contribution of Indirect Land Use Change (iLUC) to the potential Climate Change impact of bio-based insulation boards alternatives was calculated according to the approach outlined in Section 4.4.16.4 of the Method (Report I). A sensitivity analysis on the applied method has been performed, with the respective results being presented in section 9.7.7.1.

<sup>152</sup> The reference flow is the mass of building insulation panel material required to fulfil the functional unit.

1 In order to apply (recalculated) iLUC GHG emission factors from the EU 2015/1513  
2 Directive (EC, 2015), the specific land demand for the production of relevant crops  
3 ( $\text{m}^2 \cdot \text{year} / \text{kg crop}$ ) was calculated first. The calculation was based on the aggregated  
4 amount of arable and agriculture land occupation flows reported, for the relevant region  
5 (here EU), in the dataset used to model the production of the specific crop. If the  
6 geography of such flows was not specified, all the arable and agricultural land occupation  
7 flows reported in the dataset were aggregated. These estimates were checked against  
8 values of land demand calculated based on 5-years average crop yields from FAOSTAT  
9 (FAO, 2019), which were found to be generally aligned with the former. Hence, the  
10 values estimated based on land occupation flows were applied, for consistency with the  
11 overall LCI modelling of scenarios.

12 Specific land demand for crop production was then converted into a demand per  
13 functional unit, based on the specific consumption of crop for polymer production ( $\text{kg}$   
14  $\text{crop/kg polymer}$ , consistently with the values applied in the LCI modelling) and the  
15 amount of polymer needed to fulfil the functional unit (reference flow) in the relevant  
16 scenario. The iLUC contribution to the Climate Change impact was finally calculated by  
17 applying to the latter the recalculated GHG emission factors from the Directive. The  
18 calculation steps to estimate the iLUC contribution to the potential Climate Change  
19 impact are summarised in Table 9.5.

20

1                    **Table 9.5.** Calculation of the iLUC contribution to the potential Climate Change impact of the bio-PUR insulation board scenario.

Scenario/ Polymer	Feedstock	Land demand for crop production <sup>(1)</sup> [m <sup>2</sup> *y/kg <sub>crop</sub> ]	Crop demand for polymer production <sup>(2)</sup> [kg <sub>crop</sub> /kg <sub>polymer</sub> ]	Polymer demand per FU <sup>(3)</sup> [kg <sub>polymer</sub> /FU]	iLUC GHG emission factor [kg CO <sub>2</sub> eq./ (m <sup>2</sup> *y)]	iLUC climate change impact [kg CO <sub>2</sub> eq./FU]
S6 - Bio- based PUR	Soybean (EU-28)	2.1 (5.5)	1.04 (1.95)	1.91 (4.9)	0.204	0.86

(1) 2.1 m2\*y/kg soybean oil after market-based allocation (5.5 m2\*y/kg soybean); based on arable land occupation exchanges reported in the dataset.  
(2) 1.04 kg soybean oil/kg polyol (1.95 kg soybean per kg polyol).  
(3) Reference flow is 4.9 kg PUR per FU; there are 0.39 kg polyols from soybean oil per kg PUR, i.e. polyol reference flow=4.9\*0.39=1.91 kg polyol/FU.



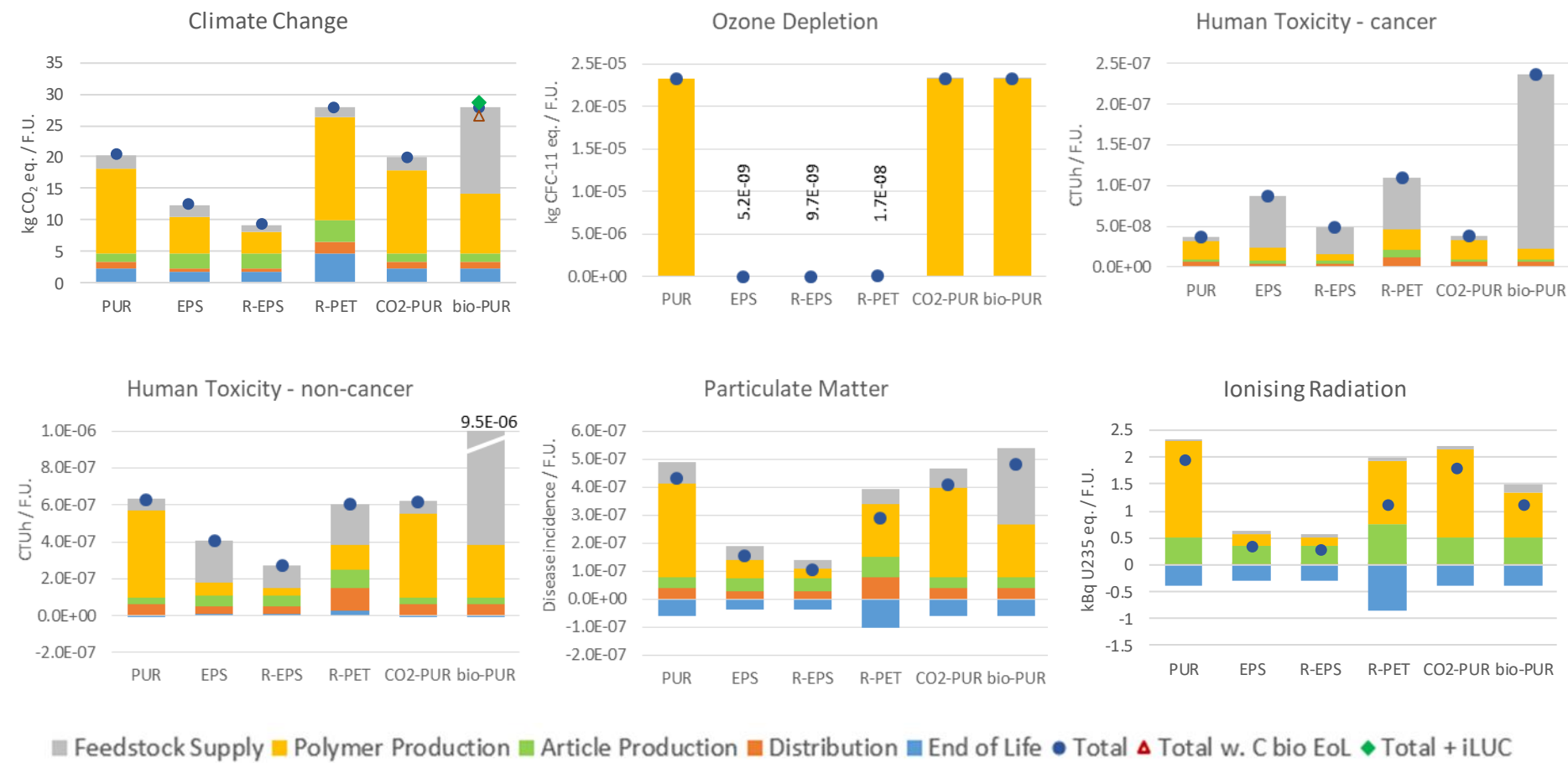
## 9.5 Life Cycle Impact Assessment results

The characterised potential impacts of the scenarios assessed are reported in Figures 9.7 to 9.9. These also show the breakdown of contributions from the main lifecycle stages, which include:

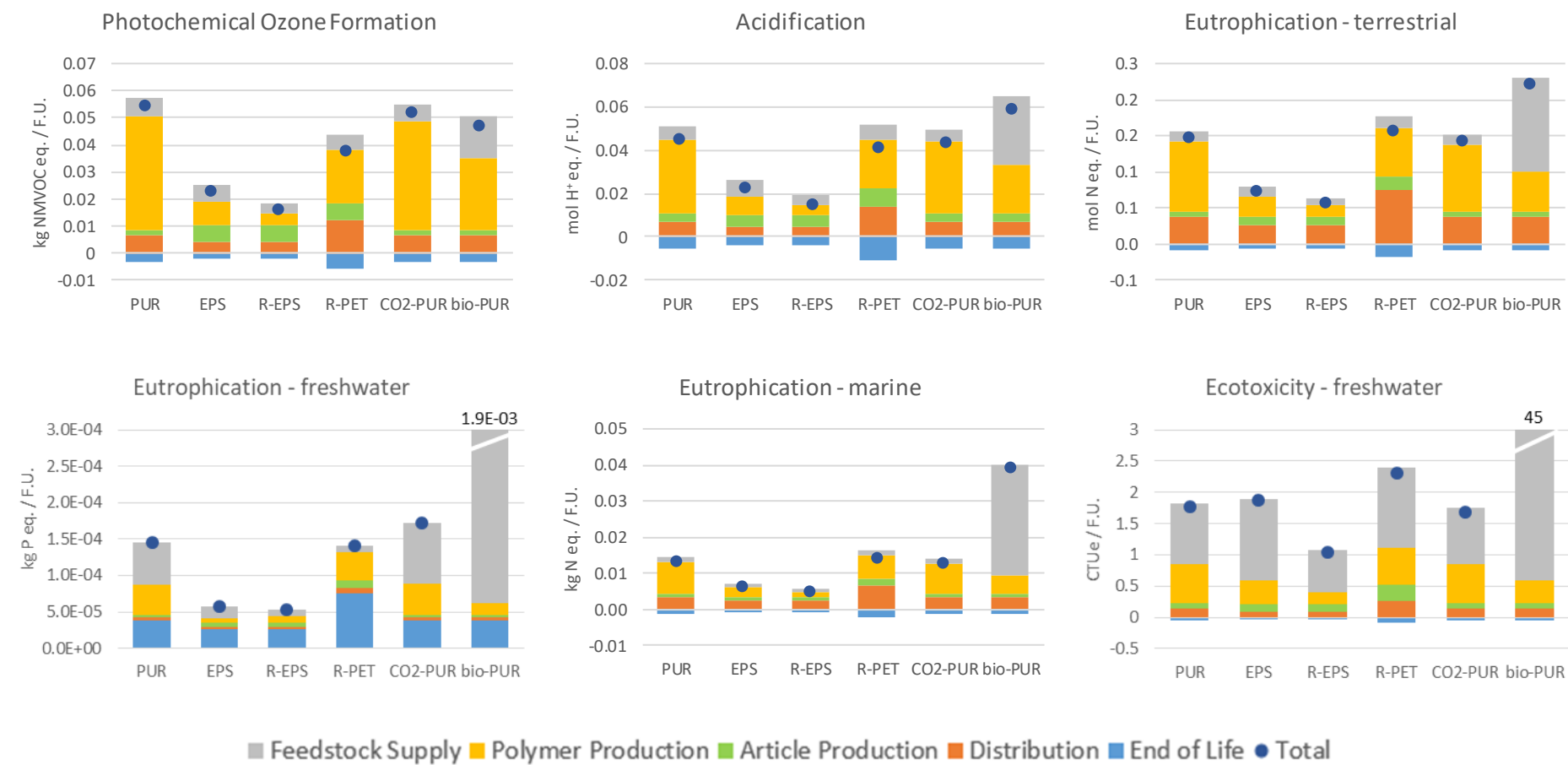
- i) *Feedstock Supply*, i.e. oil/natural gas extraction, transport and possible refining, transport of naphtha from refinery to downstream users; collection, transport and sorting of post-consumer plastic waste (recycled polymers); crop cultivation and transport to further processing (bio-based polymers); or CO<sub>2</sub> capture and its transport via pipeline to subsequent utilisation;
- ii) *Polymer Production*, i.e. all gate-to-gate activities from feedstock processing to monomer/intermediate production and polymerisation, including any transport among these, as well as transport of polymer granulate to the insulation board manufacturing site;
- iii) *Article Production*, i.e. conversion of the polymer into the finished article, i.e. expansion, cutting into boards and packaging;
- iv) *Distribution*, of the insulation boards from the manufacturing site to the final user; and
- v) *End of Life*, i.e. waste collection, transport and treatment or disposal, as well as any avoided processes from downstream displacement of virgin materials and energy.

Figure 9.10 shows the contribution of each sub-category to the Climate Change impact (i.e. from fossil, biogenic and land use/land transformation GHG emissions), while the overall characterised, normalised and weighted impact assessment results are available in Annex B.6. The latter are not reported in this section to keep readability of the document, despite this is not in alignment with the position suggested in the LCA report template provided in the underlying Method (Report I).

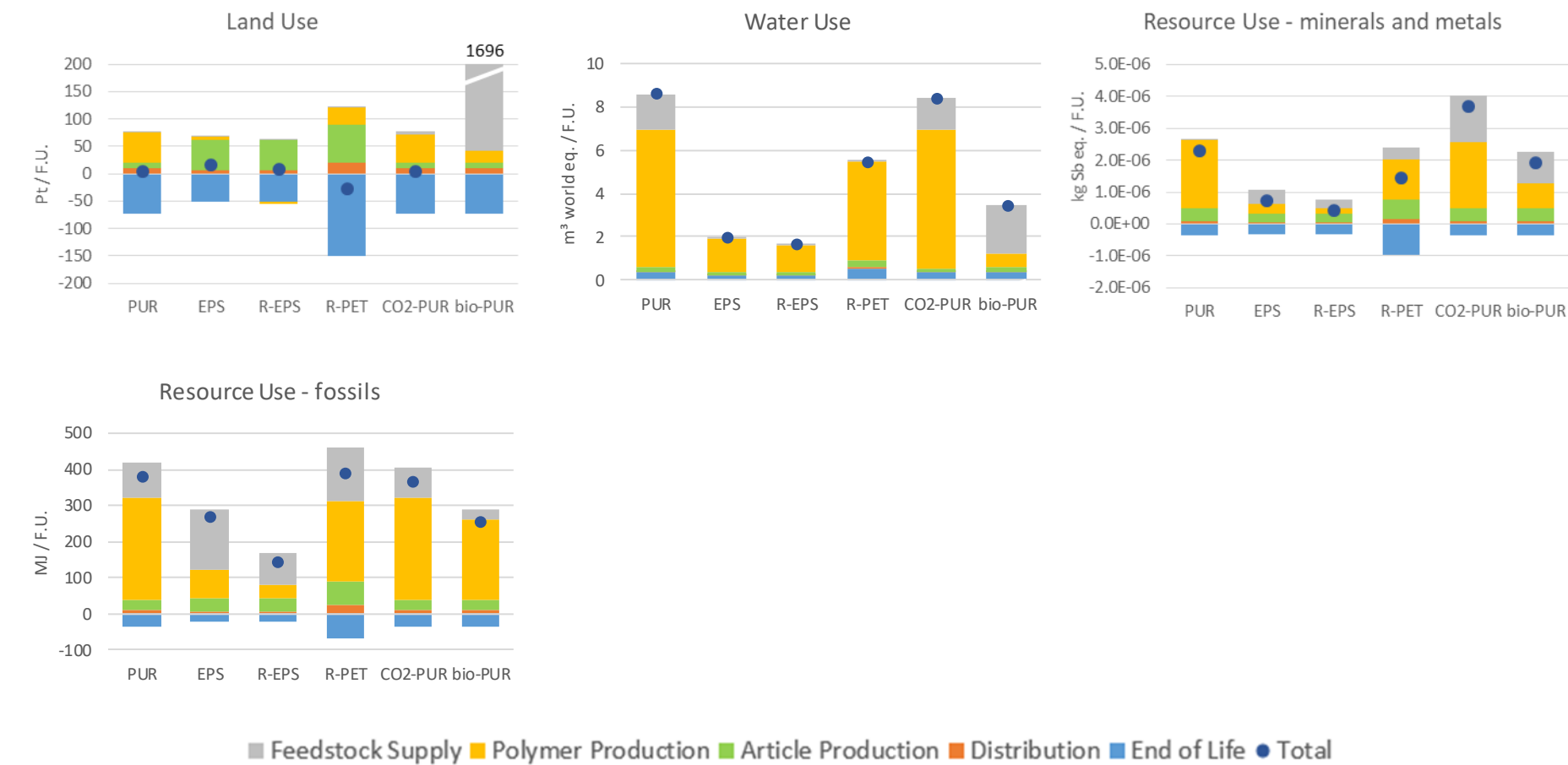
Note that scenario impacts presented in Figures 9.7 to 9.9 refer to the EU-average End of Life scenario (as described in Section 9.4.5.1), and represent the net impacts from waste management activities, resulting from the balance between real burdens and benefits (if any). Potential impacts calculated assuming 100% insulation boards being routed to each viable End of Life option are presented in Figures 9.11 to 9.13. We warn that this analysis should not be interpreted as a direct LCA comparison of alternative End of Life management options for insulation boards, because a product-based perspective is considered and allocations among consecutive product life cycles are applied to the recycling pathway. Conversely, in a typical waste management LCA that compares alternative End of Life options (i.e. with a functional unit of 1 tonne waste managed), the recycling pathway would be assigned the full burdens and benefits it involves, as there is no need to break the system mass flows between the system recycling and the one using the recycled feedstock (i.e. system perspective; no allocation needed).



**Figure 9.7.** Potential impact of insulation boards LCA scenarios for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity - non-cancer, Particulate Matter and Ionising Radiation. Note that in some impact categories a part of the results is out of scale and is curtailed. Climate Change impacts denoted with "C bio EoL" accounts for the contribution of biogenic carbon not released after 100 years from landfilling of bio-based insulation boards.

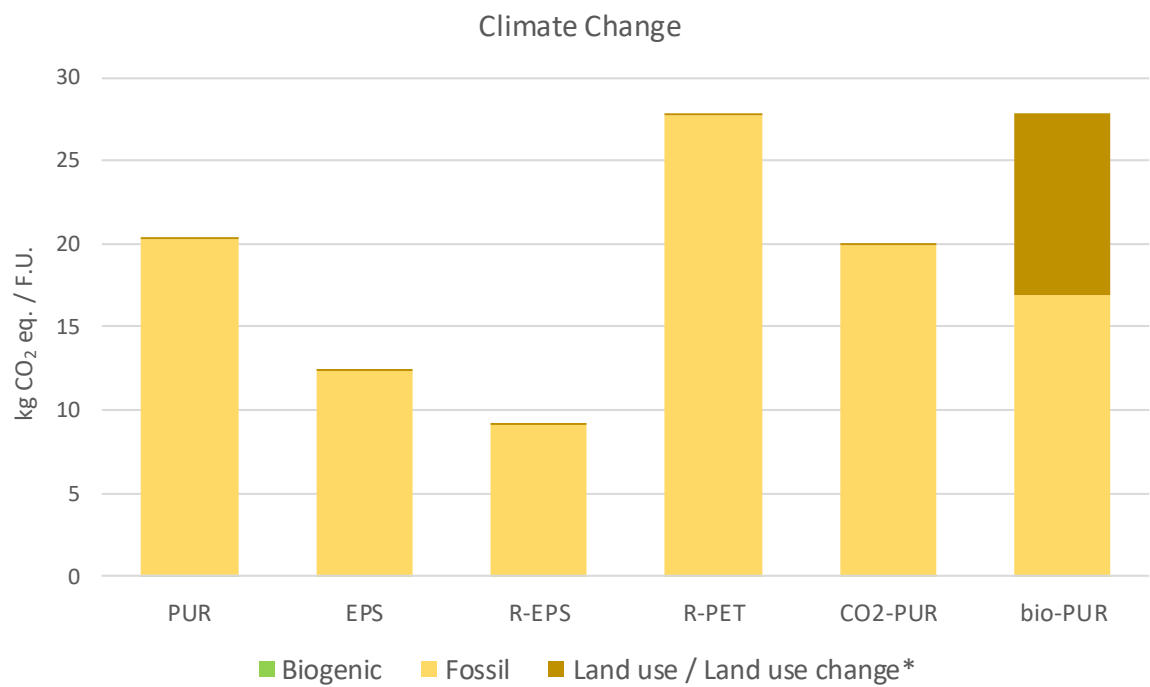


**Figure 9.8.** Potential impact of insulation boards LCA scenarios for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater. Note that in some impact categories a part of the results is out of scale and is curtailed.



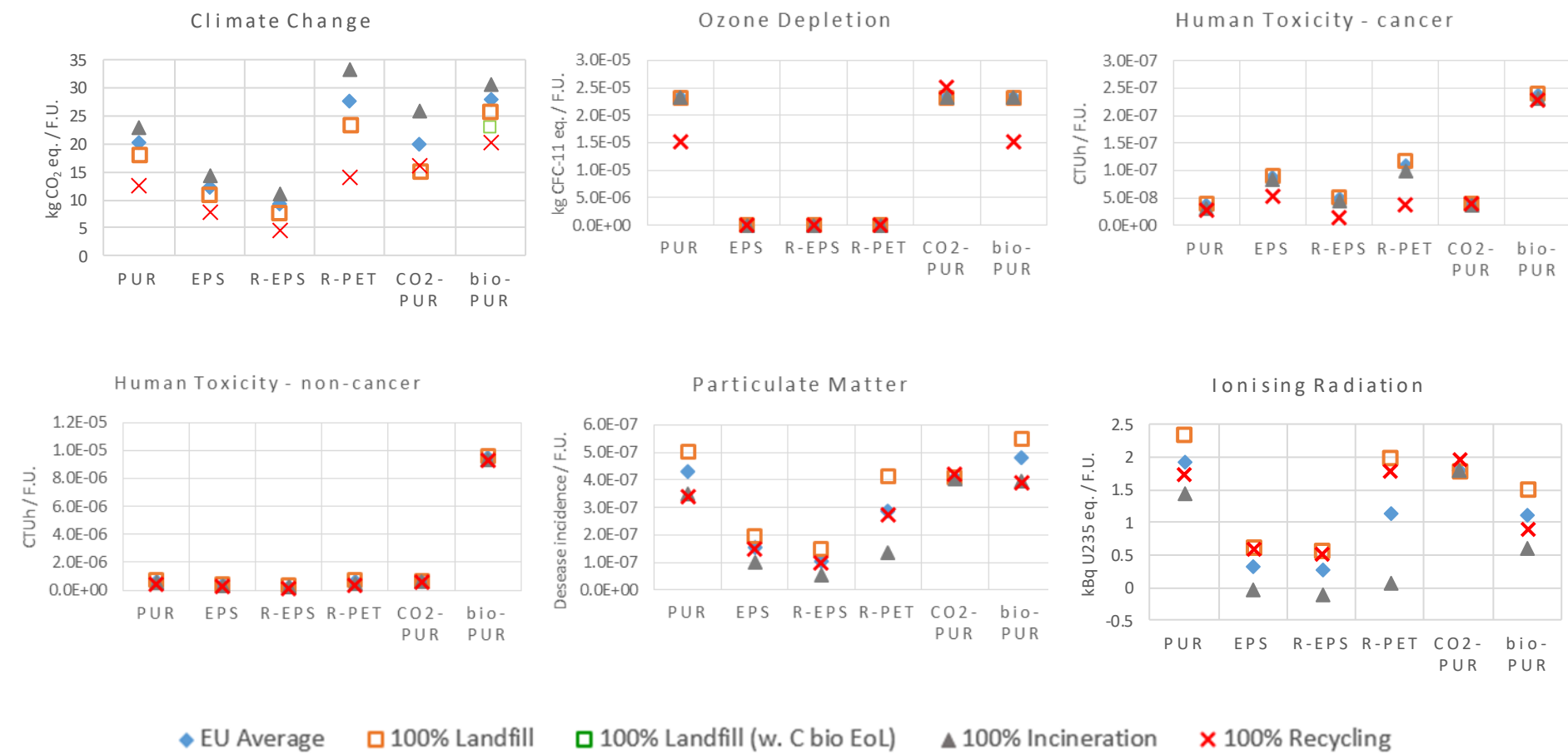
**Figure 9.9.** Potential impact of insulation boards LCA scenarios for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils. Note that in some impact categories a part of the results is out of scale and is curtailed.

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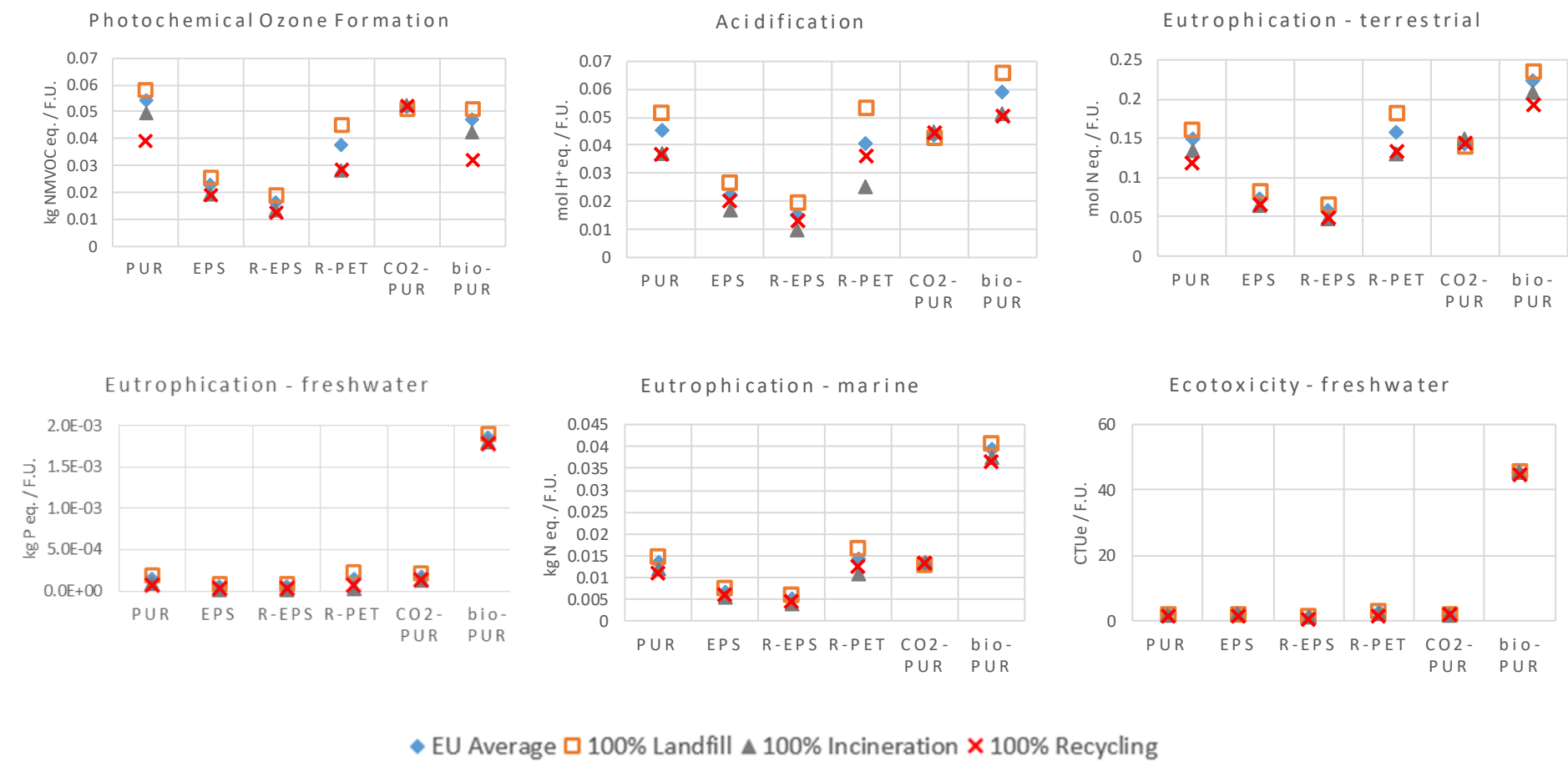


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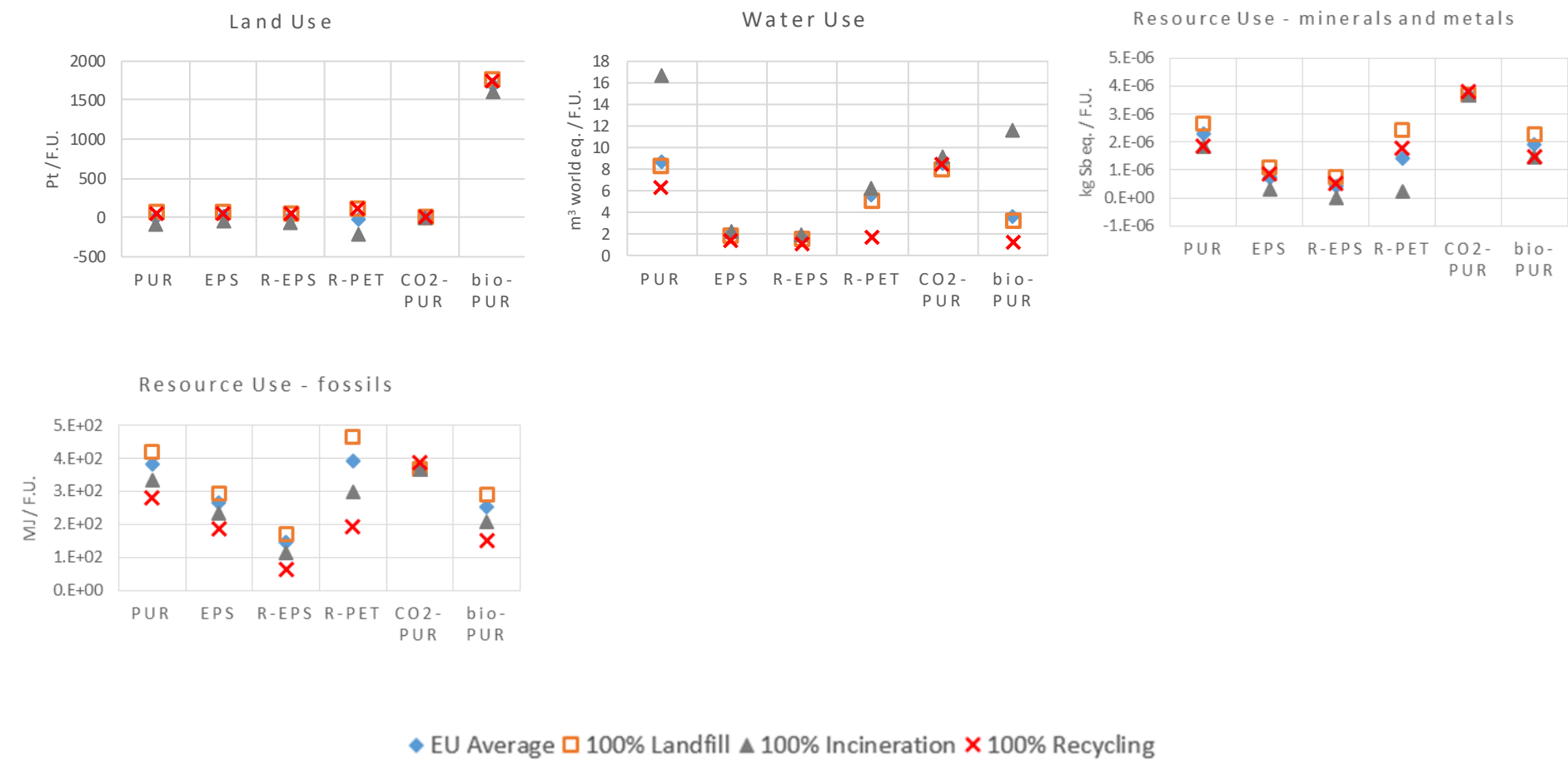
**Figure 9.10.** Contribution of fossil, biogenic, and land use/land transformation GHG emissions to the total Climate Change impact of insulation boards LCA scenarios (note that the contribution of iLUC to land transformation emissions is excluded).



**Figure 9.11.** Potential impact of insulation boards LCA scenarios for different End of Life options, for the categories of Climate Change (without iLUC), Ozone Depletion, Human Toxicity - cancer, Human Toxicity – non-cancer, Particulate Matter and Ionising Radiation. Climate Change impacts of the 100% landfilling scenario denoted with “C bio EoL” accounts for the contribution of biogenic carbon not released after 100 years from landfill deposition of bio-based insulation boards.



**Figure 9.12.** Potential impact of insulation boards LCA scenarios for different End of Life options, for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater.



1 **Figure 9.13.** Potential impact of insulation boards LCA scenarios for different End of Life options, for the categories of Land Use, Water Use, Resource  
2 Use - minerals and metals, Resource Use – fossils.



## 9.6 Additional environmental information

This section presents additional environmental impacts or aspects going beyond the default impact categories considered in the present method, which reflect official categories adopted in the PEF context. Additional environmental issues addressed include indirect Land Use Change (iLUC) impact on Climate Change, potential Biodiversity impacts, as well as the contribution of the investigated insulation boards scenarios to macro-plastics formation at End of Life (including product litter) and to micro-plastics generation throughout the supply chain.

### 9.6.1 iLUC impacts

Table 9.6 presents the estimated contribution of GHG emissions from iLUC induced by bio-based feedstock supply to the potential Climate Change impact of bio-based insulation boards.

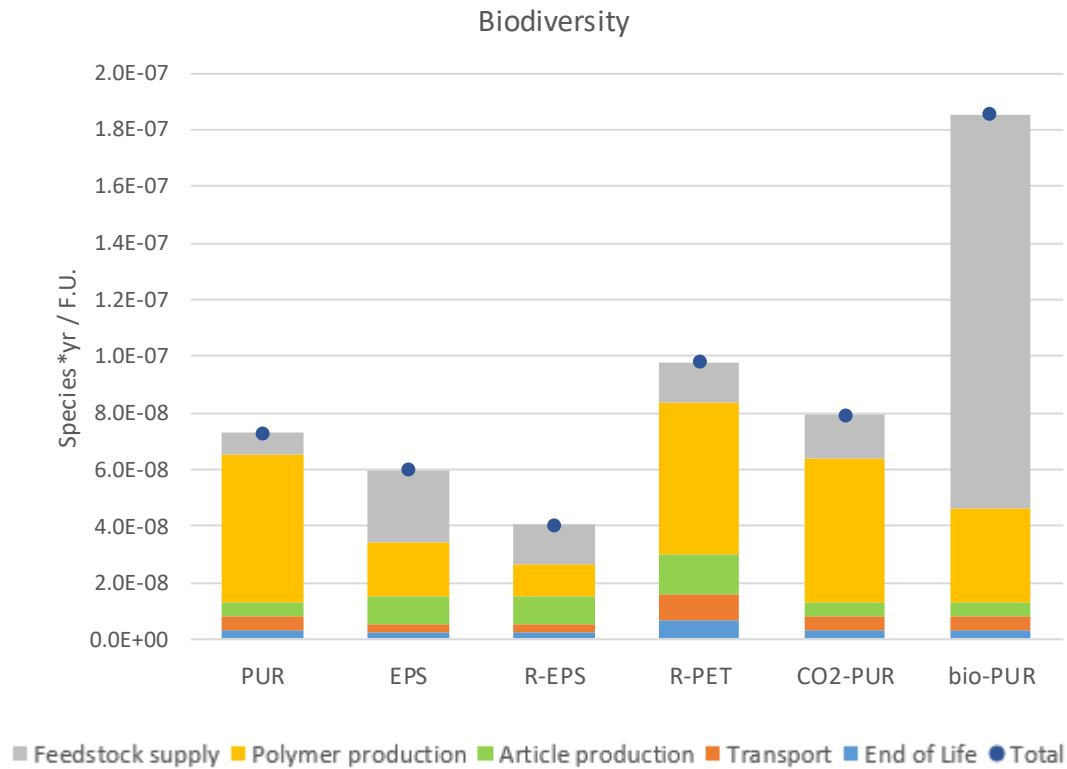
**Table 9.6.** iLUC contribution to the potential climate change impact of the insulation boards LCA scenarios.

Scenario/ Polymer	iLUC Climate Change impact [kg CO <sub>2</sub> eq/FU]	Total Climate Change impact (incl. iLUC) <sup>(1)</sup> [kg CO <sub>2</sub> eq/FU]
S1 – PUR	-	(20.3)
S2 – EPS	-	(12.4)
S3 – R-EPS	-	(9.1)
S4 – R-PET	-	(27.8)
S5 – CO <sub>2</sub> -PUR	-	(20.0)
S6 – Bio-PUR	0.86	28.7 (27.9)

<sup>(1)</sup> Values in parenthesis refer to the Climate Change impact without the iLUC contribution.

### 9.6.2 Biodiversity impacts

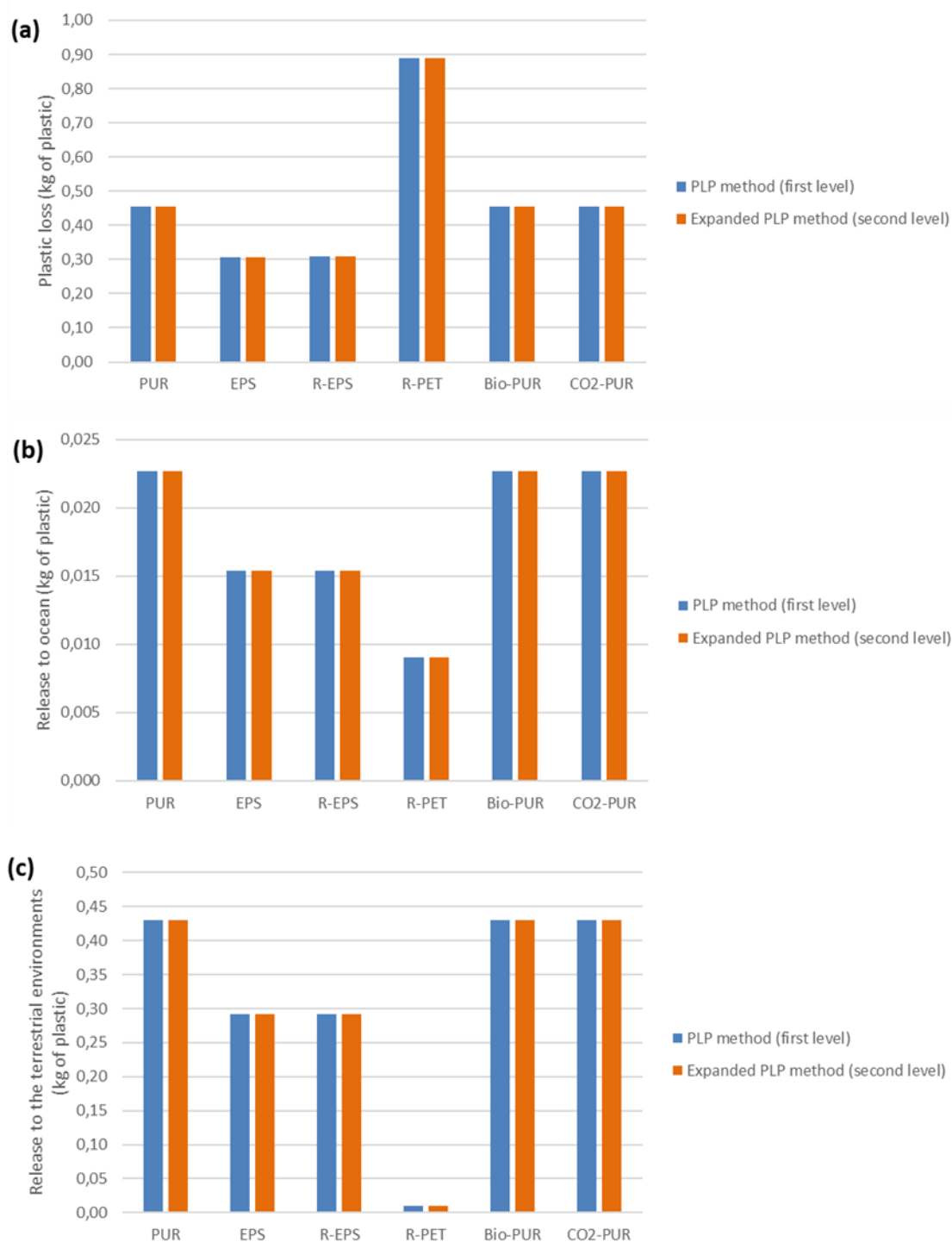
Estimated potential Biodiversity impacts of the investigated scenarios are presented in Figure 9.14. It is reminded that the assessment is performed through an endpoint-level indicator relying on impact assessment methods that for some of the underlying midpoint impact drivers differ from those recommended in the present method.



**Figure 9.14.** Potential biodiversity impact of insulation boards LCA scenarios, expressed as potential loss of species per year.

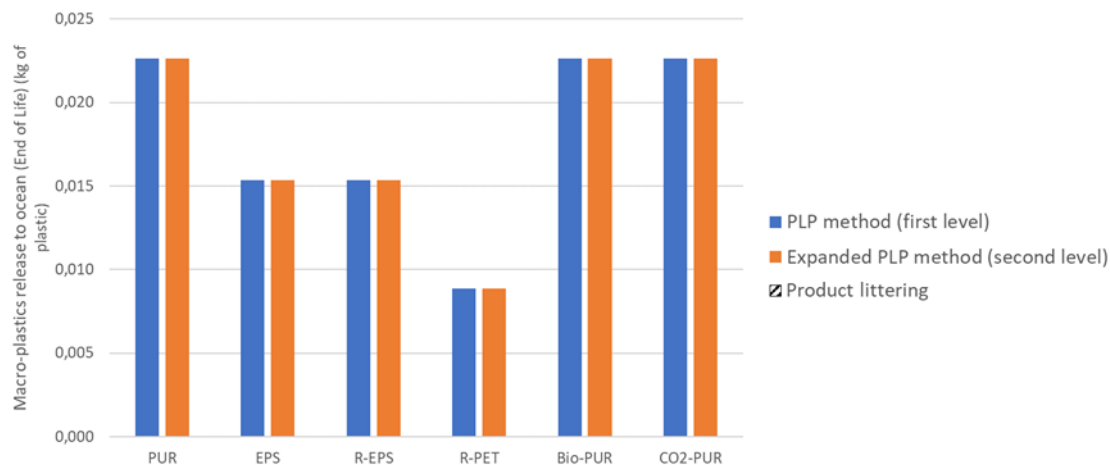
### 9.6.3 Macro- and micro-plastics generation

Figure 9.15 shows the total plastic loss, release to ocean and release to the terrestrial environment estimated for the assessed building insulation boards scenarios, considering the whole product life cycle. Estimates obtained by applying all the two different explored approaches are presented, including the total loss and release to ocean/terrestrial environment calculated through both the PLP method (first level) and the Expanded PLP method (second level).



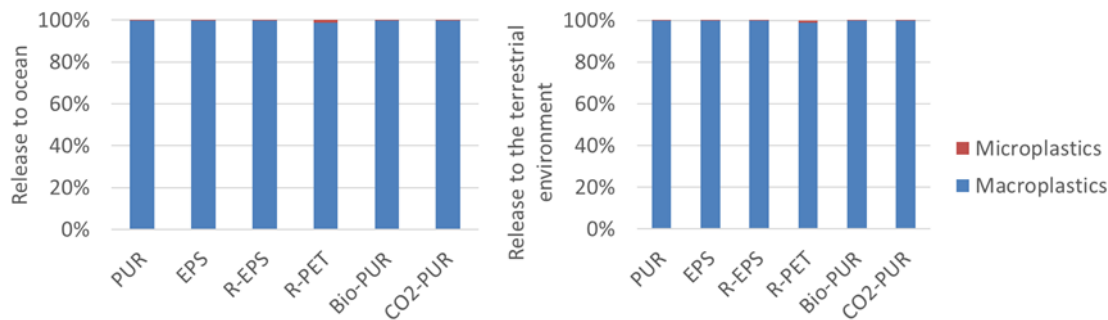
**Figure 9.15.** Total plastic loss (a), release to ocean (b) and release to the terrestrial environment (c) estimated for building insulation boards LCA scenarios with different approaches. The contribution of both macro- and micro-plastics is included, considering the whole product life cycle.

Regarding the sole release of macro-plastics to ocean at End of Life (from product littering and waste mismanagement), Figure 9.16 presents the contribution of each building insulation boards scenario, estimated with the two different explored approaches: the PLP method (first level), the Expanded PLP method (second level).



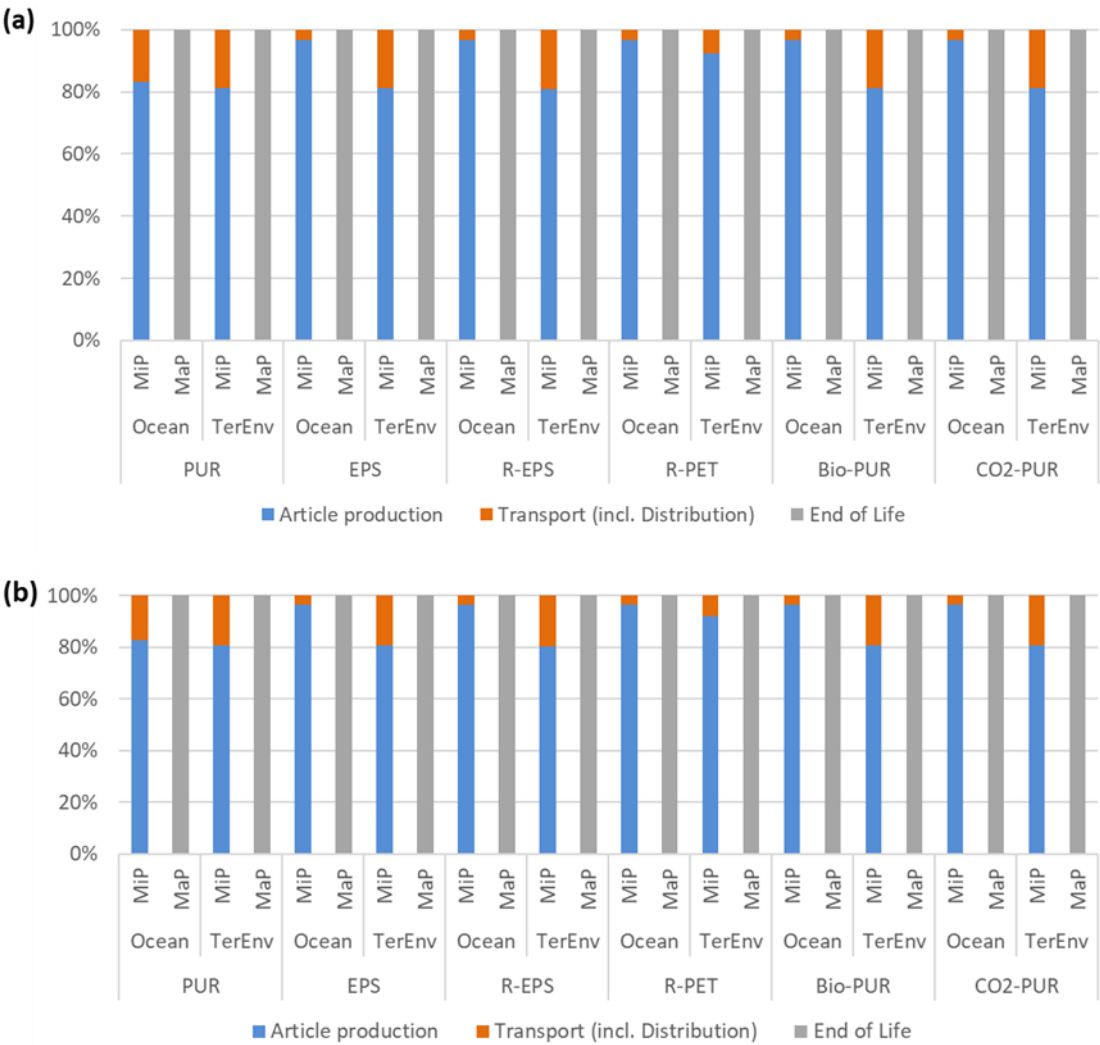
**Figure 9.16.** Macro-plastics release to ocean at End of Life, estimated with different approaches.

As for the share between the release of macro- and micro-plastics, Figure 9.17 shows as an example the relative contribution of the two categories to the total release to ocean and to the terrestrial environment estimated through the PLP method (first level). Note that the results are similar to those obtained when applying the Expanded PLP method (second level), as further discussed in Section 9.7.7.



**Figure 9.17.** Contribution of macro- and micro-plastics to the total plastic release to ocean and to the terrestrial environment estimated with the PLP method for each building insulation boards LCA scenario.

The contribution of the different lifecycle stages to the total macro- and micro-plastics release to ocean and to the terrestrial environment of the assessed building insulation boards scenarios has been finally evaluated. The results of this “hotspot analysis” are reported in Figure 9.18 for both the PLP method and the Expanded PLP method. Results are not shown for the other (sensitivity) approaches as they either cover only plastic losses or only a part of the product life cycle (i.e. End of Life).



**Figure 9.18.** Contribution of life cycle stages to the total macro-plastics (MaP) and micro-plastics (MiP) release to ocean (Ocean) and to the terrestrial environment (TerEnv) estimated for building insulation boards LCA scenarios with (a) the PLP method (first level) and (b) the Expanded PLP method (second level).

## 9.7 Interpretation

In the interpretation of the case study results, the most relevant impact categories and life-cycle stages of the studied systems are first identified (9.7.1 and 9.7.2, respectively). Scenario impacts are then compared, considering both characterised LCIA results (9.7.3) and total normalised and weighted impacts (9.7.4). Results obtained by applying individually each viable End of Life option are separately discussed in Section 9.7.6, while the effects of iLUC and the contribution to macro- and micro-plastics generation are assessed in Sections 9.7.5 and 9.7.7. Finally, the results of the sensitivity analyses on three relevant methodological choices are presented in Section 9.7.8.

Most relevant processes were not identified, since it was not possible to achieve a similar level of vertical disaggregation for foreground processes included in the life cycle models of the analysed scenarios. Therefore, the identification of most relevant processes cannot be carried out consistently across all scenarios, and a more detailed investigation for specific scenarios is not meaningful. The identification of most relevant elementary flows was also not undertaken, as this requires the identification of most relevant processes. Note, however, that any company, organisation or any other supply chain actor applying

the present method shall proceed with the identification of both most relevant processes and elementary flows.

### 9.7.1 Identification of most relevant impact categories

Table 9.7 shows the most relevant impact categories identified for each insulation boards scenario, based on normalised and weighted impacts, according to the approach outlined in the Method (Report I – Section 6.2.1). Relevant categories were identified as those that cumulatively contribute to at least 80% of the total normalised and weighted impact of the considered impact categories, excluding toxicity-related ones (these being still based on characterisation factors implemented in the EF 2.0 impact assessment methods, and hence not yet updated based on REACH data<sup>153</sup>). 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 (Section 6.2.1 of the Method (Report I)).

For all the case studies considered, Climate Change and Resource Use – fossils were identified as relevant impact categories. For the specific case of fossil-based PUR, CO<sub>2</sub>-based PUR and bio-based PUR insulation boards, also Photochemical Ozone Formation, Water Use, Ozone Depletion, Particulate Matter, and Land Use were identified as relevant in order to reach the impact threshold considered in the methodology (80%).

**Table 9.7.** Most relevant impact categories identified for insulation boards LCA scenarios and related contribution to the total normalised and weighted scenario impact.

S1 – Fossil-based PUR		S2 – Fossil-based EPS		S3 - Recycled EPS	
Impact category	Contrib.	Impact category	Contrib.	Impact category	Contrib.
Climate Change	39%	Resource Use – fossils	43.3%	Climate Change	47.6%
Resource Use – fossils	34.8%	Climate Change	42%	Resource Use – fossils	35.9%
Water Use	4.5%	Photochemical Ozone Formation	3.4	Photochemical Ozone Formation	3.7
Photochemical Ozone Formation	4.5%				
<b>Total</b>	<b>82.7%</b>	<b>Total</b>	<b>88.6%</b>	<b>Total</b>	<b>87.2%</b>
S4 – Recycled PET		S5 – CO <sub>2</sub> -based PUR		S6 – Bio-based PUR	
Impact category	Contrib.	Impact category	Contrib.	Impact category	Contrib.
Climate Change	50.4%	Resource Use – fossils	39.1%	Climate Change	47.5%
Resource Use – fossils	33.8%	Climate Change	34.3%	Resource Use - fossils	20.6%
Acidification	3.1%	Water Use	4.6%	Land Use	6.4%
		Ozone Depletion	4.5%	Particulate Matter	4.3%
				Acidification	4.2%
<b>Total</b>	<b>87.3%</b>	<b>Total</b>	<b>82.6%</b>	<b>Total</b>	<b>81.3%</b>

<sup>153</sup> According to the latest version of the Product Environmental Footprint Category Rules Guidance, toxicity-related impact indicators based on these characterization factors shall be excluded from the procedure to identify most relevant impact categories.

## 9.7.2 Identification of most relevant life-cycle stages

Table 9.8 shows the most relevant lifecycle stages in the relevant impact categories identified in Section 9.7.1, and the associated contribution, quantified according to the rules detailed in the Method (Report I – Section 6.2.2). Most relevant stages include those that together contribute to at least 80% of the total characterised lifecycle impact in the specific category and are highlighted in yellow in Table 9.8. For a more exhaustive picture, the contribution of all lifecycle stages is reported.

While Table 9.7 is self-explanatory, an explanation of the specific processes contributing to the impact for each relevant life cycle stage may be found in section 9.7.3. Note that for PUR (conventional and CO<sub>2</sub>-based), the life cycle stage “Feedstock Supply” only includes the provision of the feedstock to produce polyols while the supply of the remaining fossil chemicals needed to produce the final polymer are included under the stage “Polymer Production”.

**Table 9.8.** Contribution of life cycle stages to the total characterised impacts of the most relevant categories identified for insulation boards LCA scenarios. Most relevant stages (i.e. those contributing to more than 80% of the total impact) are highlighted in yellow.

S1 – Fossil-based PUR					
Climate Change		Resource Use - fossils		Water Use	
Life cycle stage	Contrib.	Life cycle stage	Contrib.	Life cycle stage	Contrib.
Polymer Production	66.7%	Polymer Production	74.6%	Polymer Production	73.7%
End of Life	11.8%	Feedstock Supply	24.6%	Feedstock Supply	19.7%
Feedstock Supply	10.3%	Article Production	7.2%	End of Life	4.0%
Article Production	6.8%	Distribution	3.1%	Article Production	2.3%
Distribution	4.3%	End of Life	-9.5%	Distribution	0.2%
Photochemical Ozone Formation					
Life cycle stage	Contrib.				
Polymer Production	77.5%				
Feedstock Supply	12.6%				
Distribution	11.6%				
Article Production	4.0%				
End of Life	-5.8%				
S2 – Fossil-based EPS					
Resource Use - fossils		Climate Change			
Life cycle stage	Contrib.	Life cycle stage	Contrib.		
Feedstock Supply	63.6%	Polymer Production	47.9%		
Polymer production	29.6%	Article Production	19.3%		
Article Production	12.7%	Feedstock Supply	14.6%		
Distribution	3.0%	End of Life	13.4%		
End of Life	-9.1%	Distribution	4.8%		

1

S3 – Recycled EPS					
Climate Change		Resource Use - fossils			
Life cycle stage	Contrib.	Life cycle stage	Contrib.		
Polymer Production	38.3%	Feedstock Supply	60.3%		
Article Production	26.1%	Polymer Production	27.7%		
End of Life	18.1%	Article Production	23.4%		
Feedstock Supply	11%	Distribution	5.5%		
Distribution	6.5%	End of Life	-16.9%		
S4 – Recycled PET					
Climate Change		Resource Use - fossils			
Life cycle stage	Contrib.	Life cycle stage	Contrib.		
Polymer Production	59%	Polymer Production	57.7%		
End of Life	17.2%	Feedstock Supply	37.8%		
Article Production	12.6%	Article Production	16.7%		
Distribution	6.2%	Distribution	5.9%		
Feedstock Supply	5.0%	End of Life	-18.1%		
S5 – CO <sub>2</sub> -based PUR					
Resource Use - fossils		Climate Change		Water Use	
Life cycle stage	Contrib.	Life cycle stage	Contrib.	Life cycle stage	Contrib.
Polymer Production	77%	Polymer Production	66.3%	Polymer Production	76.3%
Feedstock Supply	22.2%	End of Life	12.0%	Feedstock Supply	17.4%
Article Production	7.4%	Feedstock Supply	10.4%	End of Life	4.1%
Distribution	3.2%	Article Production	6.9%	Article Production	1.9%
End of Life	-9.8%	End of Life	4.4%	Distribution	0.2%
Ozone Depletion					
Life cycle stage	Contrib.				
Polymer Production	99.8%				
Feedstock Supply	0.13%				
End of Life	0.02%				
Feedstock Supply	0.002%				
Distribution	0%				

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<b>S6 – Bio-based PUR</b>					
<b>Climate Change</b>		<b>Resource Use - fossils</b>		<b>Land Use</b>	
<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%
<b>Particulate Matter</b>		<b>Acidification</b>			
<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%		

2

### 3 9.7.3 Interpretation of characterised results

4 This section provides a comparison between the characterised impacts of insulation  
5 boards (Figures 9.7-9.9 and Table B.6.7 in Annex B.6) produced using alternative  
6 feedstock or materials and insulation boards produced from fossil sources, i.e.  
7 conventional PUR and EPS, used as references of comparison. The comparison focuses on  
8 the sixteen, default impact categories considered in the Product Environmental Footprint  
9 (PEF) context and in this assessment. Potential impacts on Biodiversity are discussed  
10 separately, as being estimated via an endpoint indicator that is not recommended in the  
11 PEF framework, and partially relying on different impact assessment methods for the  
12 underlying midpoint-level impact categories. Note that, in the following comparison,  
13 differences between scenario impacts lower than 10% were not considered significant, in  
14 light of the uncertainty associated with the developed life cycle models and the applied  
15 impact assessment models. Moreover, the any comparative considerations related to  
16 Human Toxicity and Ecotoxicity impact categories need to be interpreted with caution  
17 given the uncertainty of the underlying impact assessment models and results. The same  
18 applies, to a lower extent, also to water Use and Land Use.

19 Focusing on the two fossil-based reference polymers (i.e. PUR and EPS), the performance  
20 of EPS insulation board was superior across all the impact categories considered with the  
21 exception of Human Toxicity – cancer, where feedstock supply for EPS incurred larger  
22 impacts than that for PUR. The overall better performance is mainly explained by the  
23 reduced amount of polymer required to fulfil the desired service in the case of EPS (i.e.  
24 the reference material flow; for EPS 3.3 kg and for PUR 4.9 kg; see Table 9.3). It should  
25 also be noticed that the relatively higher impact associated with PUR on Ozone Depletion  
26 is here entirely due to MDI use in the PUR production process. Recycled EPS insulation  
27 board (scenario namely R-EPS) showed lower impacts compared with conventional EPS  
28 insulation board (and therefore with PUR) in all impact categories mainly owing to the  
29 reduced impact associated with feedstock supply and subsequent polymer production  
30 thanks to recycling.

Recycled PET insulation board (scenario namely R-PET) incurred larger impacts compared with EPS insulation board in all the impact categories assessed, except for Land Use where the performances were comparable. The overall worse performance is explained by the notably increased amount of PET-polymer required to fulfil the desired service as compared with EPS (the reference flow for PET equals 9.6 kg versus 3.3 kg for EPS; see Table 9.3). Compared with PUR insulation board, the performance of R-PET was better in Ozone Depletion, Photochemical Ozone Formation, Water Use, and Resource Use – minerals and metals and worse in Climate Change and Human Toxicity – cancer. In the remaining categories the performances of the two scenarios appeared comparable. The performance of CO<sub>2</sub>-based PUR insulation board (CO<sub>2</sub>-PUR) was better than fossil-based PUR in the categories Climate Change and Photochemical Ozone Formation while comparable in the remaining categories. When compared with EPS, the performance of this scenario was however worse in all the impact categories assessed.

The performance of bio-based PUR insulation board (Bio-PUR) was worse than EPS in all the impact categories assessed but Resource Use – fossils, due to the lower demand for fossil fuel. The reason for the worse performance was bi-fold: i) the increased reference material flow required to fulfil the functional unit (4.9 kg versus 3.3 kg; see Table 9.3) and ii) the increased burden from feedstock supply, i.e. emissions associated with crop production practices, and polymer production, i.e. oil refining operations and subsequent processing of the refined soy oil into polyol. In respect to feedstock supply, agricultural production was the main responsible for the impacts incurred in Eutrophication (both marine and freshwater), Land Use, Human Toxicity (cancer and non-cancer) and Ecotoxicity following use of fertilisers and pest-control for cropping. When compared with PUR, Bio-PUR achieved a better result in Resource Use – fossils, Water Use, Ionizing Radiation, and Photochemical Ozone Formation due to decreased burdens from feedstock supply and polymer production.

#### **9.7.4 Interpretation of normalised and weighted results**

This section provides a comparison between the normalised and weighted results of the scenarios investigated. Excluding the toxicity categories from the weighted score calculation (Table B.6.9 in Annex B.6), recycled EPS insulation board (R-EPS) achieved the best score followed by conventional EPS, CO<sub>2</sub>-based PUR (CO<sub>2</sub>-PUR), conventional PUR, recycled PET (R-PET) and Bio-PUR insulation boards.

#### **9.7.5 Effects of indirect land use change (iLUC)**

When the contribution of GHG emissions from iLUC is taken into account, the burden on Climate Change is increased only by about 3% for the bio-based PUR insulation board (Figure 9.7 and Table 9.5). On this basis, accounting for iLUC effects does not affect the relative performance of the scenarios assessed in this case study (i.e. their relative comparison is not affected). However, this outcome needs to be read also in light of the GHG emission factors applied for iLUC assessment, which appear to fall in the lower end of the range of values available in the recent literature (see Section 4.4.16.3 in the Method (Report I)). The use of an alternative iLUC model and of the resulting emission factors was explored as a sensitivity analysis (see Section 9.7.7.1).

#### **9.7.6 Alternative End of Life options**

This section discusses characterised scenario results calculated by individually applying each viable End of Life option as an exclusive (100%) End of Life scenario (as presented in Figures 9.11-9.13). The main purpose is to evaluate how the performances of the single insulation board scenarios are affected by the change in the applied End of Life option, rather than to mutually compare such scenarios for different End of Life options. However, the results should not be interpreted as a strict comparison between different End of Life alternatives for the studied article, since the evaluation applies to a product perspective, and burdens/benefits of selected End of Life options (e.g. recycling) are partitioned between different product life cycles (which prevents from capturing the full

implications of having a given waste stream routed to a specific End of Life option). In a waste management system perspective, where allocation is not needed, the total (system-wise) savings associated with the EoL pathway “100% recycling” as such would likely be higher than what presented in this report, where a product-perspective is applied.

None of the three considered product End of Life options (i.e. 100% mechanical recycling, 100% incineration and 100% landfilling) could be identified as preferred across all the impact categories and scenarios assessed. However, for all scenarios the product recycling option stood out as the best option in Climate Change, Resource Use – fossils, Photochemical Ozone Formation, Human Toxicity - cancer and Water Use; this is expected because of the avoided extraction of fossil fuel for virgin production (saving on resources) and of the avoided incineration of plastic waste (saving on air emissions such as CO<sub>2</sub> and NO<sub>x</sub>). Recycling was also the best EoL option in Ozone Depletion for all the scenarios involving PUR production because of the decreased use of chemicals (e.g. MDI) in the polymer processing stage.

Incineration was the best option for EPS, R-EPS, and R-PET insulation boards in Particulate Matter, Ionising Radiation, Acidification, and Resource Use – minerals and metals. The favourable performance was mainly related to the savings obtained from substitution of the average EU energy production mix (both electricity and heat) and to the decreased collection and sorting effort required compared with recycling that requires implementation of separate collection systems and sorting.

In most of the investigated impact categories, landfilling was the worst option confirming to a large extent the priorities outlined in the “Waste Hierarchy”, which sets disposal as the least preferable option (EC, 2008). This was however not the case in Climate Change and Water Use. For the first, carbon from landfilling of these (non-biodegradable) materials remains indeed largely non-degraded during the 100y time horizon considered, i.e. stored, thus avoiding CO<sub>2</sub> release. For the second, less industrial processing is required overall lowering water consumption compared with incineration.

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 these environmental compartments are typically affected by emissions of metals and nutrients (e.g. Zn, Cr, Cd, Cu, Ni, N and P) after material application on agricultural land, which does not apply to this case study (no compost or digestate is involved in the scenarios). Likewise, comparable impacts were observed in the category Land Use.

### **9.7.7 Interpretation of results related to macro- and micro-plastics generation**

This section discusses the results presented in Section 9.6.3 on the estimated potential generation of macro- and micro-plastics of the assessed building insulation boards scenarios. The latter are compared according to the same criteria applied to the interpretation of the potential environmental impacts (Section 9.7.3), initially focusing on the results from the two-level approach applied as a base case. No sensitivity analyses were assessed for this case study.

Regarding the total loss of plastics (macro- and micro-plastics) and the resulting release to ocean and to the terrestrial environment (Figure 9.15), the two base-case approaches (PLP method and Expanded PLP method) provide almost exact results (<0.01% difference) due to the lack of a recycling at the End of Life of the product and the potential over-mismanagement of waste in non-EU countries importing plastic waste for recycling.

When comparing the different building insulation boards scenarios, plastic loss indicates that best alternatives are EPS and R-EPS (-33% compared to PUR). At the plastic loss level, R-PET results in the worst scenario (+96% when compared to PUR), as it requires

the largest amount of polymer. These results are mostly a consequence of the amount of polymer required per functional unit, which leads to a proportionally increased or decreased loss of plastics from both upstream life cycle stages (microplastics) and especially from product End of Life (macro-plastics). Note, however, that the applied methods rely on mass-based indicators (which are affected by differences in board mass) rather than quantifying the number of items (or plastic fragments/particles) lost. If this was the case, all scenarios would result in the same macro-plastics generation at End of Life when applying both PLP method and Expanded PLP method, since boards with a lower mass would not imply a reduction in the contribution of product littering and waste mismanagement (which are based on release rates that are not affected by the type of feedstock or material). In turn, an identical macro-plastics generation at End of Life would imply having a comparable total loss, these being mostly determined by macro-plastics (see below).

Conversely, when considering the release to the environment, R-PET becomes the scenario with the lowest released amount to both ocean and the terrestrial environment. This is related to the modelling parameters considered for the release rates in the PLP method, which are based on the economic value of the material and its recovery as resource input to the technosphere. PET is considered a material with high value leading to a release rate of 1% for both environmental compartments. This is a very low rate compared to the other scenarios, which assume a release rate to the ocean of 5% and a release rate to the terrestrial environment of 95%. This highlights the high sensitivity of the results to the defined parameters.

As shown in Figure 9.17 for the PLP method, the total plastic release to both ocean and the terrestrial environment is dominated by the release of macro-plastics at the End of Life (more than 99%), while micro-plastics released throughout the upstream life cycle have only a minor role. Similar results are also observed for the Expanded PLP method (not shown), despite two additional sources of micro-plastics are considered in the latter (i.e. road markings and marine coatings). In both cases, this is a consequence of the prevailing mass of macro-plastics, which is directly related to the mass of boards required per functional unit (reference flow), by means of higher loss and release rates compared to micro-plastics. Therefore, the mass of macro-plastics is at least one order of magnitude higher than the mass of micro-plastics, which instead depends on the quantity of relevant lifecycle processes by means of (much) lower loss and release coefficients (depending on the source).

With regard to the origin of released plastics (in terms of life cycle stage and source), macro-plastics entirely originate from the End of Life stage (Figure 9.18), as they are only derived from mismanagement of the product as waste. Note that no littering is assumed for building insulation boards as it is considered that boards are managed as part of buildings at the End of Life thereby following specific collection and management pathways. On micro-plastics release, the relative contribution of the different sources (e.g. loss of plastic pellets and tire abrasion) depends on the characteristics of the supply-chain (e.g. feedstock origin, which affect the contribution from transport across the life cycle), as well as on the mass of polymer required per functional unit (which affect, to different extents, all the considered sources). However, within the limited role of micro-plastics in the overall release to the environment, the most relevant contribution is generally provided by pellet losses from the Polymer and Article Production stages, while transport-related sources have a more restricted role.

In conclusion, the lowest release of macro- and micro-plastics to the environment (both ocean and terrestrial) is associated to the R-PET scenario, mainly associated to modelling parameters derived from the economic value of the material. However, the use of EPS and R-EPS are the scenarios showing the lowest plastic loss due to the dominant role of macro-plastics loss at the End of Life of building insulation boards, which associates the best overall performance to those scenario requiring a lower amount of plastic to fulfil the functional unit (i.e. relying on lighter building insulation boards; mass-driven results).

From a methodological perspective, the assumption of an End of Life without a recycling pathway leads to strongly similar results between the different approaches.

### 9.7.8 Sensitivity analysis

A sensitivity analysis has been performed on a number of relevant parameters or assumptions, to evaluate the effects of their variation on the potential impacts of the affected scenario(s), and on the comparative LCA results. The following aspects have been specifically considered:

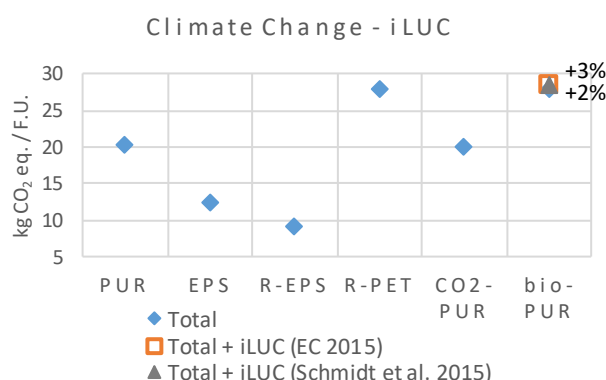
1. iLUC model and factors.
2. Modelling of CO<sub>2</sub> as a feedstock for polyol production.
3. Accounting of temporary carbon storage and delayed emissions.

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

#### 9.7.8.1 iLUC model and factors

This sensitivity analysis explores the use of an alternative model (i.e. Schmidt et al., 2015) to quantify the iLUC contribution to the Climate Change impact of bio-based insulation boards (i.e. Bio-PUR). The iLUC factors calculated through the mentioned model were applied in place of the GHG emission factors from EC (2015) applied as a base case (Table 9.6). The results are displayed in Figure 9.19, limited to the Climate Change category, which is the only one affected by this sensitivity analysis.

Overall, the application of alternative iLUC factors only marginally affects the results. The impact increase observed when applying EC (2015) factors was equal to 3% in Bio-PUR insulation boards, while applying the model from Schmidt et al. (2015) leads to a comparable increase of 2%. The reason for this is that the iLUC factor specific for oil-based crops used for the baseline is higher than the iLUC GHG factor calculated using the model by Schmidt et al. (2015). It should be noticed, however, that only a minor share of the PUR is composed of carbon derived from biomass (ca. 16%). This partly explains the relatively low iLUC contribution. In conclusion, the results of this case study can be considered reasonably robust with respect to the estimated iLUC contribution (which is negligible), and the comparison among the affected insulation boards scenarios and the respective fossil-based scenarios of reference is not affected by the application of alternative factors.



**Figure 9.19.** Results of the sensitivity analysis on the applied model to quantify the iLUC contribution to the total Climate Change impact of bio-based insulation boards.

#### **9.7.8.2 Modelling of CO<sub>2</sub> as a feedstock for polyol production**

This sensitivity analysis explores the application of a number of alternative methodological approaches to address the burden of CO<sub>2</sub> used as a feedstock for polyol production (in scenario #5; CO<sub>2</sub>-PUR). In particular, the following approaches (described more in detail in Section 4.4.4 of the Method (Report I) and summarised as well in Section 5.7.8.6 of this report) were assessed:

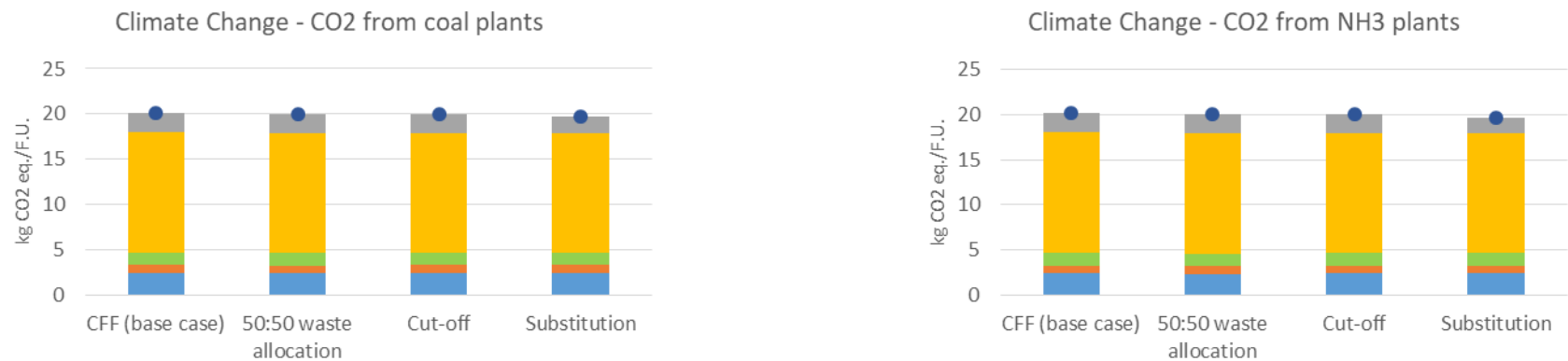
- i) “Cut-off” approach (raw gaseous CO<sub>2</sub> as a waste for recycling);
- ii) System Expansion via substitution (CO<sub>2</sub>-based polyols as a co-product of the whole Carbon Capture and Utilisation system).
- iii) “50:50” waste allocation approach (raw gaseous CO<sub>2</sub> as a waste for recycling);

These approaches are compared against the Circular Footprint Formula (CFF), used as default approach in this study.

Results are displayed in Figure 9.20. The three approaches are compared with the baseline, where the CFF approach is applied. The sensitivity analysis is carried out for the case of CO<sub>2</sub> feedstock captured from both coal-fired power plants and NH<sub>3</sub> producing plant, i.e. from natural gas as a C fossil source. For the purpose of clarity, only the Climate Change category is addressed. The difference between the individual approaches in this case study is very limited because the amount of fossil carbon captured (as fossil CO<sub>2</sub>-C) only constitutes a minor share of the total carbon in the PUR (ca. 3%) and therefore of the total dry matter (ca. 1.8%). This largely explains the reduced differences across approaches. Notice also that in Climate Change the only difference between the cut-off approach and system expansion via substitution lies in that system expansion credits the scenario for capturing the CO<sub>2</sub> that would be otherwise released to air (in the substituted, uncoupled process of electricity or ammonia production). This means that, compared with the cut-off approach, the fossil CO<sub>2</sub> not emitted (i.e. the share of CO<sub>2</sub>-C embodied in the product) contributes to the Climate Change impact with -1 kg CO<sub>2</sub>-eq/kg CO<sub>2</sub> not released<sup>154</sup>.

All in all, applying the 50:50 waste allocation and the CFF led to a slight increase of the impact on Climate Change as compared to the cut-off approach, because the scenario was assigned a portion of the burdens from virgin polyol production (CFF approach) or from fossil fuels extraction and supply (50:50 waste allocation). While coal and natural gas supply certainly incur different impacts, the difference between the two CO<sub>2</sub> sources was here negligible, as the amount of fossil carbon captured was minor, as explained earlier. Overall, the CFF appeared to be the approach that scored the highest impact albeit the magnitude was nevertheless comparable to the remaining approaches. The results for the remaining categories, other than Climate Change, did not significantly vary across the individual approaches applied (not presented).

<sup>154</sup> Like any other saved (i.e. negative) fossil CO<sub>2</sub> emission, which according to the applied (EF 2.0) LCIA method, is assigned a characterisation factor equal to 1 kg CO<sub>2</sub>-eq./kg CO<sub>2</sub>.



Life Cycle stage	CFF (base case)	50:50 waste allocation	Cut-off	Substi-tution
Feedstock supply	2.09	2.11	2.08	1.78
Polymer production	13.38	13.25	13.25	13.25
Article production	1.39	1.39	1.39	1.39
Distribution	0.87	0.87	0.87	0.87
End of Life	2.40	2.35	2.40	2.40
Total	20.13	19.97	19.99	19.69

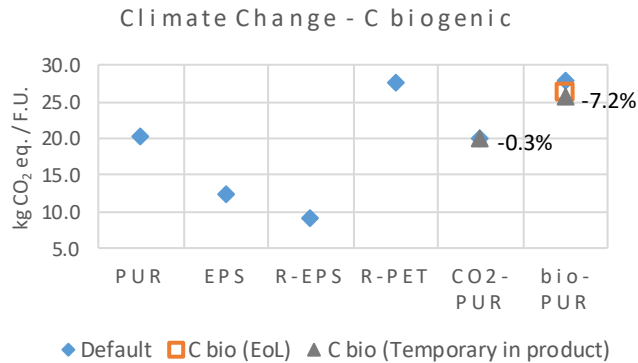
Life cycle stage	CFF (base case)	50:50 waste allocation	Cut-off	Substi-tution
Feedstock supply	2.09	2.11	2.08	1.78
Polymer production	13.38	13.25	13.25	13.25
Article production	1.39	1.39	1.39	1.39
Distribution	0.87	0.87	0.87	0.87
End of Life	2.40	2.35	2.40	2.40
Total	20.13	19.97	19.99	19.69

■ Feedstock Supply ■ Polymer Production ■ Article Production ■ Distribution ■ End of Life ● Total

**Figure 9.20.** Results of the sensitivity analysis on the three alternative approaches to address the burden of the CO<sub>2</sub> feedstock, as compared to the approach used as default in this study (CFF: Circular Footprint Formula).

9.7.8.3 Temporary carbon storage and delayed emissions

This sensitivity analysis illustrates the GHG contribution of accounting for the temporary storage of biogenic carbon (in the case of bio-based PUR) and/or the delayed emission of fossil carbon (e.g. in the case of fossil CO<sub>2</sub> capture), as detailed in the Method (Report I). Results are displayed in Figure 9.21. Overall, crediting temporary CO<sub>2</sub> storage or delayed CO<sub>2</sub> emissions decreased the total impact on Climate Change by 1.4% for CO<sub>2</sub>-PUR and by ca. 1.9% for bio-PUR. It should be noticed that the amount of captured C in CO<sub>2</sub>-PUR equalled 0.09 kg C per FU (ca. 16.5% of polyol is composed of captured CO<sub>2</sub>; i.e. ca. 1.8% of PUR dry mass is composed of captured C), while that of biogenic C in bio-PUR equalled 0.76 kg C per FU (ca. 16% of bio-based PUR is composed of carbon derived from biomass). Considering that 55% of the total carbon is assumed to be landfilled at the EoL (99% of which being stored and never emitted), only about 45% of the captured carbon should be assigned additional credits from temporary storage or delayed emissions, as the remaining 55% is already accounted as carbon not emitted, see C bio (EoL). This means that landfilled carbon is accounted using a characterisation factor of 0 kg CO<sub>2</sub>-eq/kg CO<sub>2</sub> for the carbon in CO<sub>2</sub>-PUR and of -1 kg CO<sub>2</sub>-eq/kg CO<sub>2</sub> for the carbon in bio-PUR; 1 kg C=3.67 kg CO<sub>2</sub>. In other words, only a minor share of the total carbon in the material is actually eligible for (additional) credits for temporary carbon storage or delayed emission credits, which partly explains the minor GHG savings achieved.



**Figure 9.21.** Quantification of benefits in Climate Change from temporary carbon storage and delayed CO<sub>2</sub> emissions. C bio = Carbon biogenic. EoL = End of Life.



1    **10 Case study 7: Automotive interior panels**

2    This case study focuses on automotive interior panels, covering an area of 1 m<sup>2</sup> of car  
3    door with a thickness of 0.8 mm. The 0.8 mm thickness was chosen as a common value  
4    typically found in the automotive interior panel industry (Benecke-Kaliko AG, 2018).

5    **10.1 Assessed scenarios**

6    The use of different materials and/or feedstocks for automotive interior panel  
7    manufacturing was explored by assessing six alternative scenarios, as shown in

Table 10.3. Three reference scenarios based on fossil-based plastics, namely fossil PP, fossil ABS and fossil PBS, were analysed (scenarios 1, 2 and 3). In addition to this, the use of recycled PP from separately collected, post-consumer plastic waste as a feedstock was also explored (R-PP, Scenario 4), assuming a 100% recycled content. Although different shares of recycled material can be mixed with virgin material to be used as input to panel production, this study focuses on panels made entirely of recycled input. This is justified by the fact that car panels are not “sensitive” plastic products such as food contact packaging items, where the amount of some pollutants must be strictly limited. Additionally, by considering 100% recycled content, it is possible to assess the effects of a complete substitution of the virgin material.

Two bio-based products were also assessed; bio-based PBS and PLA (scenarios 5 and 6). These were considered to be fully bio-based, i.e. all building blocks that compose the polymer are made from biological raw materials. Once again, this allows assessment of the effects of a complete fossil raw material substitution. In the case of Bio-PBS the building blocks were bio-derived succinic acid and 1,4-butanediol while PLA was considered to be produced via the fermentation of corn-derived dextrose followed by polymerisation of lactic acid. PLA is aerobically biodegradable under controlled composting conditions, and, to a lower extent, under specific anaerobic conditions typical of bio-gasification plants. PLA-based panel assessed in Scenario 5 were considered to be made from corn grown in the United States as a primary feedstock. Both fossil PBS and Bio-PBS are biodegradable according to EN 13432 (PTT MCC Biochem Co., 2014). Bio-PBS panel assessed in Scenario 6 were produced from corn grown in the United States as a primary feedstock. Fossil PBS panel assessed in Scenario 3 was produced using fossil-based succinic acid and 1,4-butanediol. Due to limitations in data availability, the results for PBS obtained in this assessment are not representative of real industrial production and should be interpreted carefully, keeping in mind that technologies with different level of maturity are compared.

1 **Table 10.3.** LCA scenarios assessed for the automotive interior panel case study.

Scenario	Polymer	Monomer/Co-polymer	Feedstock	End of Life options <sup>(1)</sup>
1 - Conventional polymer 1	PP	Propylene	Fossil-based (oil/natural gas)	Recycling Incineration Landfilling
2 - Conventional polymer 2	ABS	Acrylonitrile Butadiene Styrene	Fossil-based (oil/natural gas)	Recycling Incineration Landfilling
3 - Conventional polymer 3	PBS	Succinic acid 1,4-butanediol (BDO)	Fossil-based (oil/natural gas)	Recycling Incineration Landfilling Composting
4 - Alternative polymer 1	R-PP (100% recycled content)	Polypropylene	Waste PP (post-consumer)	Recycling Incineration Landfilling
5 - Alternative polymer 2	Bio-PBS	Bio-succinic acid Bio-BDO	Maize (USA)	Recycling Incineration Landfilling Composting
6 - Alternative polymer 3	PLA	Lactic acid	Maize (USA)	Recycling Incineration Landfilling Composting Anaerobic digestion

2 <sup>(1)</sup> The impacts of scenarios were individually assessed for each listed End of Life option, as well as for a  
3 combination of such options reflecting as far as possible the average current situation at the EU level.

## 4 10.2 Functional Unit and reference flow

5 The main function of the article under study is to cover the interior of a car's door from  
6 producers to final customers. The functional unit of this case study was defined as  
7 "covering an area of 1 m<sup>2</sup> of car door with a thickness of 0.8 mm" (Table 10.2). The 0.8  
8 mm thickness was chosen as a common value typically found in the automotive interior  
9 panel industry (Benecke-Kaliko AG, 2018).

**Table 10.2.** Definition of the functional unit for automotive interior panel LCA scenarios.

Aspect	Description
<b>“What”</b> (function(s) or service(s) provided)	Covering the interior of a car's door from producers to final customers, excluding accessories and trim parts
<b>“How much”</b> (extent of the function(s) or service(s))	An area of 1 m <sup>2</sup> of car door with a thickness of 0.8 mm
<b>“How well”</b> (expected level of quality)	Without breaking during transport and use
<b>“How long”</b> (duration/lifetime of the function or service)	During the life span of t car
<b>“Where”</b> (location/geography of the service)	In the EU

The reference flow of each scenario (i.e. the amount of panel material required in order to fulfil the functional unit), was calculated considering the density of panel for each material. Table 10.3 summarises the reference flows of auto panel in each scenario.

**Table 10.3.** Reference flow calculation for automotive interior panel LCA scenarios.

Material	Panel density (kg/m <sup>3</sup> )	Reference flow (kg/FU)
PP	900 <sup>(1)</sup>	0.72
ABS	1100 <sup>(2)</sup>	0.88
PBS	1240 <sup>(3)</sup>	0.99
R-PP	900 <sup>(1)</sup>	0.72
Bio-PBS	1240 <sup>(3)</sup>	0.99
PLA	1240 <sup>(4)</sup>	0.99

<sup>(1)</sup> (Mangaraj et al., 2009)

<sup>(2)</sup> (Omnexus, 2019)

<sup>(3)</sup> (NOVAMONT, 2020)

<sup>(4)</sup> (NatureWorks, 2019)

### 10.3 System boundary

In all scenarios, the system boundary was set in order to cover the most relevant stages and processes of the full product life cycle (cradle-to-grave perspective), as described below and depicted in Figures 10.1 to 10.6:

- Feedstock Supply<sup>155</sup>** – covering extraction, transport and possible refining of crude oil and natural gas (fossil-based polymers), collection, transport and sorting of 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);

<sup>155</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

- *Polymer Production*<sup>156</sup> – covering all the activities associated with the conversion of feedstock materials into the relevant monomer(s) and final polymer, including any transport among these activities and final transport of polymer granulate to downstream manufacturing processes;
- *Article Production*<sup>157</sup> – including panel manufacturing through injection moulding;
- *Distribution* – including transport of panels from the manufacturing site to retailers and from these to final consumers;
- *End of Life* – covering collection, transport, recycling, incineration, or disposal of the panels after use, including any avoided processes from virgin material or energy substitution.

A different nomenclature was applied for some life cycle stages compared to the default nomenclature specified in the methodological document (and in the PEF method) to make it more relevant for the investigated supply chains and the project scope. Moreover, the default “Raw Material Acquisition and Pre-processing” stage was further split into two separate sub-stages (i.e. Feedstock Supply and Polymer Production), to allow disaggregating the impacts of feedstock supply from downstream conversion processes, and hence to better appreciate any differences among the use of different feedstock sources (in line with the project goal).

Note that, in principle, to completely fulfil the functional of the study (i.e. door cover), additional items might be required (e.g. accessories, trim parts, etc.). However, given the focus of the study on a specific article (i.e. panels), these additional items are excluded from the assessment. This omission has no effects on the outcome of the comparison among the different scenarios, as it can be reasonably assumed that the same additional items are employed regardless of the material or feedstock used for panel manufacturing.

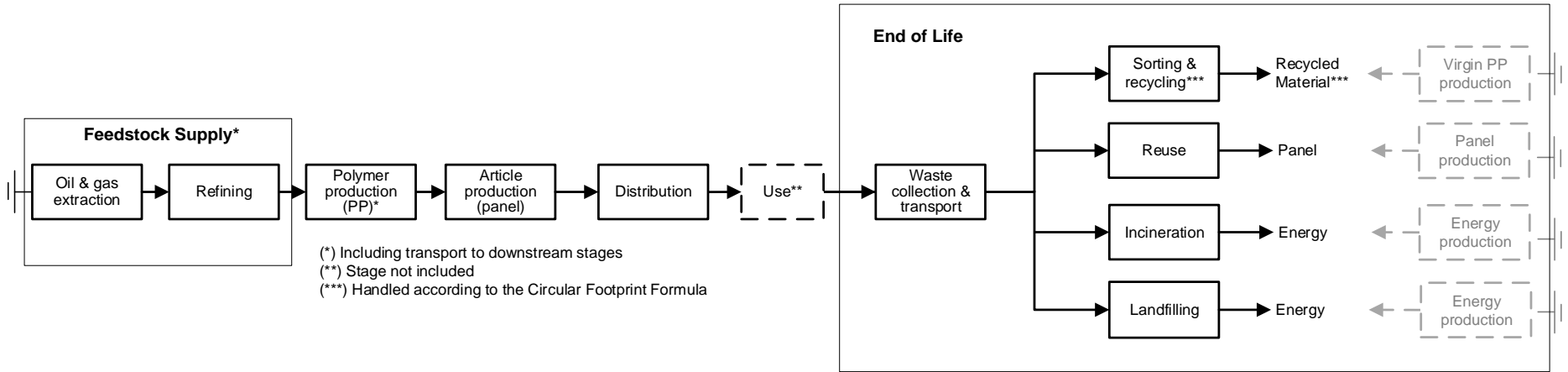
The use stage was excluded in this LCA study in order to simplify data collection. Due to the small differences in panel weight for the different scenarios and, more importantly, the low contribution of interior panels to the total vehicle kerb weight, this exclusion is not expected to have any significant effects on the comparison among different scenarios.

In general, the mentioned exclusions may weaken the assessment if they are not adequately justified (e.g. if relevant differences exist among compared alternatives, which are not taken into account). Therefore, the use stage (and the life cycle of additional accessories, trim parts, etc.) shall always be taken in full consideration for possible inclusion in a comparative LCA of automotive interior panel, paying particular attention to any differences among the compared alternatives.

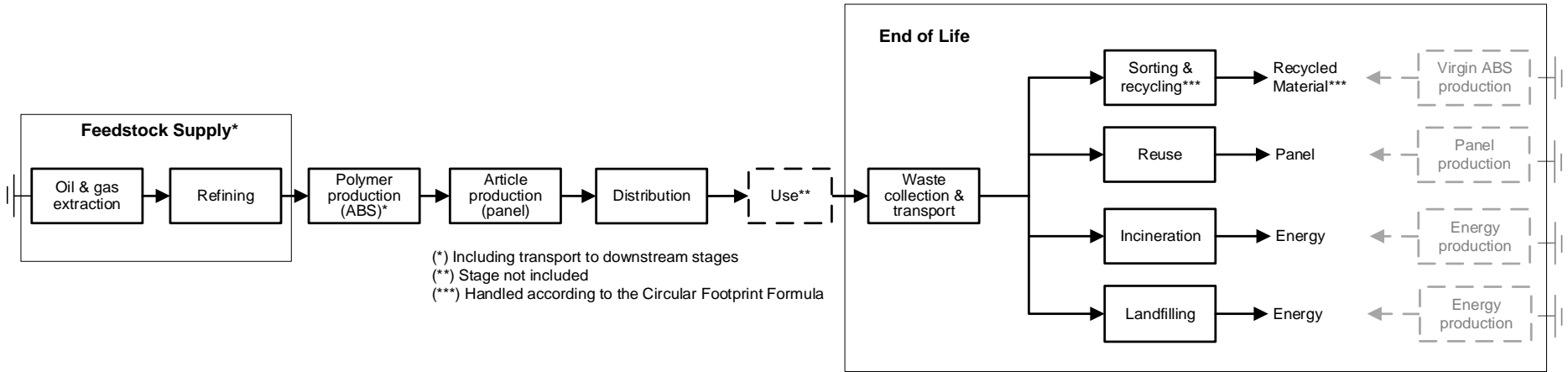
Finally, it has to be noted that additives were not included in the assessment, due to lack of complete and consistent data on the use of additives in the production of automotive panels, of the examined polymers and for plastics in general, as well as on their release and fate over the product life cycle. This is acknowledged as a limitation of this study, as additive production can account for a non-negligible portion of cradle-to-gate Climate Change impact and energy demand, which is up to 46% for (starch-based) polymer grades including larger shares of additives (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 plastics routed to biological treatments or subject to in-situ degradation).

<sup>156</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

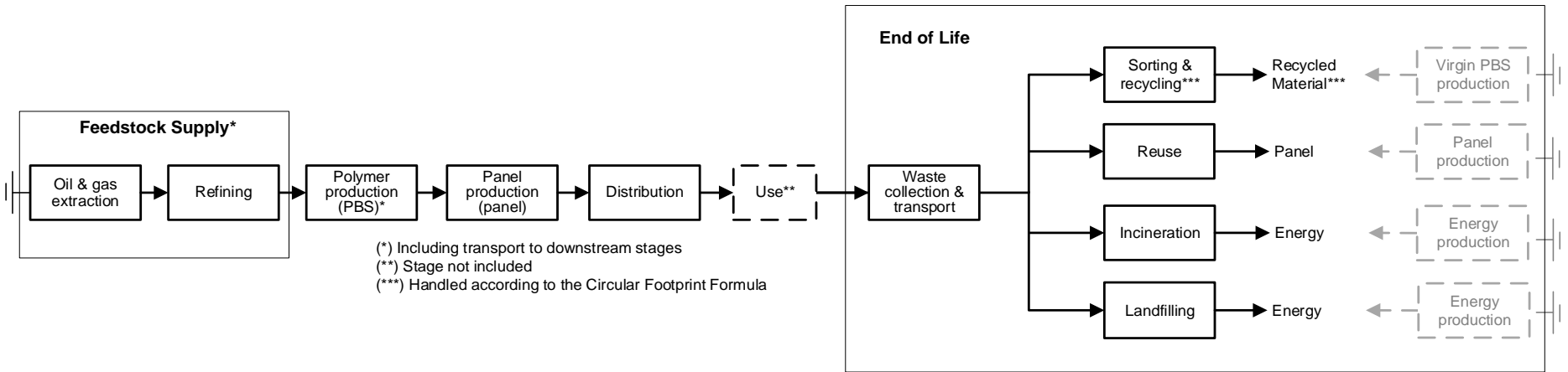
<sup>157</sup> Corresponding to the default stage of “Manufacturing” specified in the Method and in the PEF framework.



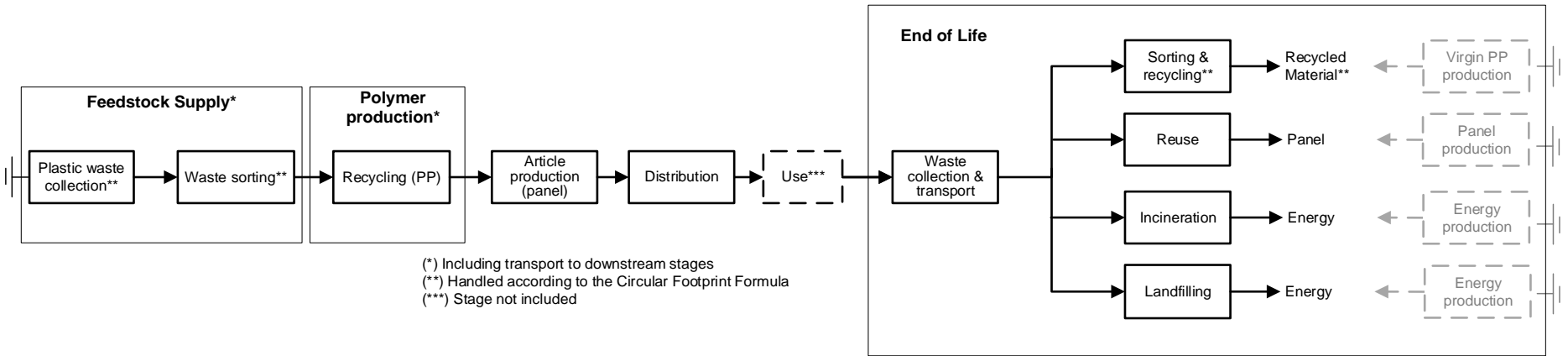
**Figure 10.3.** System boundary for fossil-based PP automotive interior panels (scenario 1).



**Figure 10.4.** System boundary for fossil-based ABS automotive interior panels (scenario 2).



**Figure 10.5.** System boundary for fossil-based PBS automotive interior panels (scenario 3).



**Figure 10.6.** System boundary for recycled PP automotive interior panels (scenario 4).

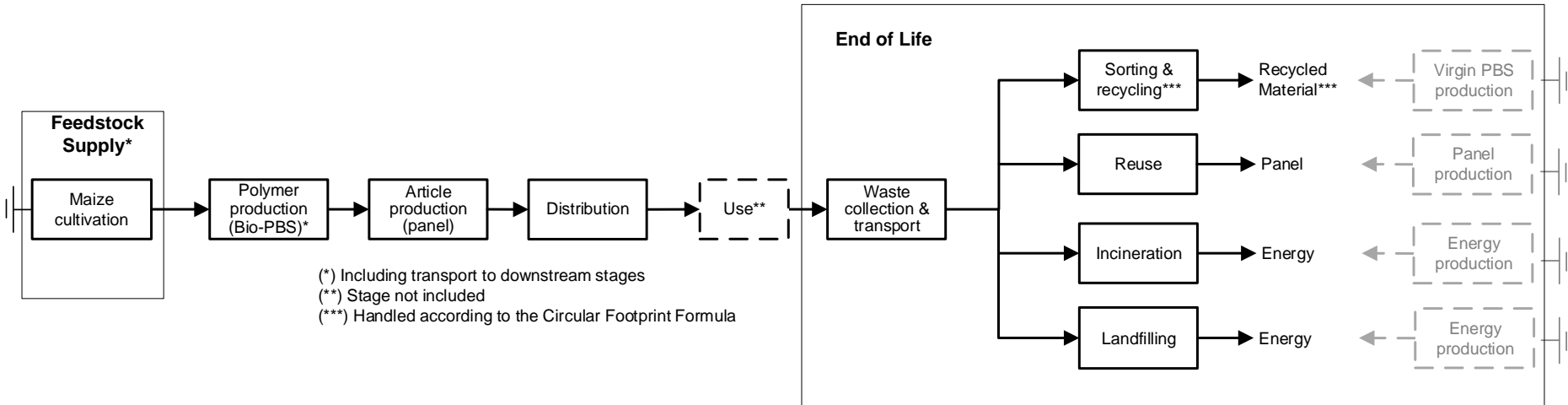


Figure 10.7. System boundary for bio-based PBS automotive interior panels (scenario 5).

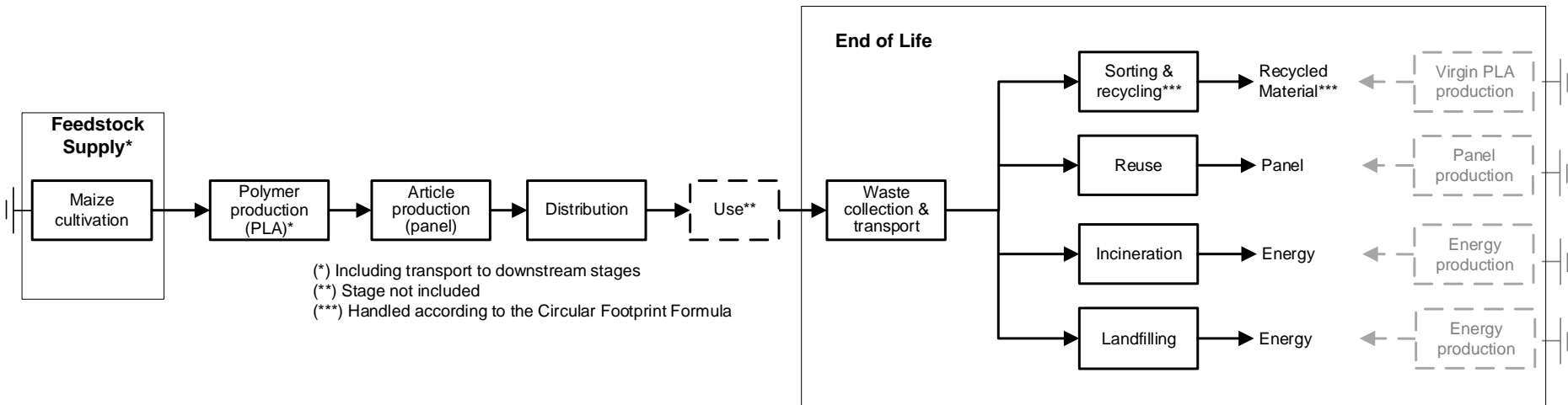


Figure 10.6. System boundary for bio-based PLA automotive interior panels (scenario 6).



## **10.4 Life Cycle Inventory**

This section describes the overall approach for developing the Life Cycle Inventory of the analysed scenarios, along with the related assumptions and data sources. The description is separated by major lifecycle stages, in the following sub-sections (10.4.1 – 10.4.4). The list of processes, related data sources, and main modelling details are provided in Tables B.7.1 to B.7.6 in Annex B7.

### **10.4.1 Feedstock Supply Stage**

#### **10.4.1.1 Fossil-based polymers**

For fossil-based polymers (PP and ABS), the stage of Feedstock Supply includes the processes of crude-oil and natural gas extraction and transport, naphtha production in crude oil refineries, and its subsequent transport to downstream conversion processes (typically naphtha cracking, or also catalytic reforming). The modelling of these activities was carried out as described in Section 4.4.1.1, where the reader is referred to for further details on data sources, covered sub-processes and related environmental burdens, main modelling choices (e.g. allocation) and any adjustments performed in the implementation of the applied datasets.

#### **10.4.1.2 Recycled polymers**

For recycled polymers (R-PP), Feedstock Supply consists of collection of post-consumer plastic waste, and its subsequent transport and sorting in specific facilities. These processes were modelled as described in Section 4.4.1.2 for the Beverage Bottle case study, and refer to collection and transport of separately collected plastic waste at the municipal level (in the absence of specific data for plastic waste collection from the different sectors it may come from), as well as to sorting of mixed plastic waste in dedicated facilities. Both processes implemented in the lifecycle model according to the Circular Footprint Formula (CFF), which is the default approach to handle recycling situations in the Product Environmental Footprint (PEF) context. According to this approach, only 50% of the burdens from collection, transport, sorting and recycling were assigned to the recycled material input used in the system ( $A=0.5$  for PP used in unspecified applications), the rest being assigned to the system providing material for recycling. Further details on the implementation of the CFF are provided in Section 4.4.2.2 on recycled polymers production.

#### **10.4.1.3 Bio-based polymers**

For bio-based polymers (Bio-PBS and PLA), the stage of Feedstock Supply includes cultivation of the relevant crop (i.e. US corn in both cases) and its subsequent transport to further processing in the same country.

Cultivation of US Maize for PLA production was modelled through an EF-compliant dataset. The latter is based on the approach used to develop agricultural inventories in the Agri-footprint database, relying on 5-year average yield data from FAOStat (2010-2014). Activities covered in the dataset are those of seeding and seed production, fertilizer and pesticide production and application, capital goods, as well as energy use and transport for field management practices. Maize was assumed to be transported to downstream processing (i.e. wet milling) along an overall distance of 100 km, covered by large lorries (> 32 t, fuelled with US diesel mix).

### **10.4.2 Polymer Production Stage**

The Polymer Production stage covers the activities of feedstock processing into relevant intermediates and monomer(s), the polymerisation process, as well as any transport among these activities and final transport of polymer granulate to the automotive panel

manufacturing site. The following subsections (10.4.2.1 – 10.4.2.4) describe how these activities have been modelled in the present case study.

#### **10.4.2.1 Fossil-based polymers**

For conventional, fossil-based polymers (PP and ABS), the whole process chain from feedstock processing to polymerisation, through the production of intermediates and monomers, was modelled by means of aggregated, gate-to-gate datasets provided by Thinkstep. Inputs includes combinations of crude oil, natural gas and naphtha, depending on the polymer. All conversion processes are assumed to take place in Europe, so that the datasets reflect EU average background conditions in terms of e.g. energy generation, material supply and transport. For both polymers, one of the main conversion processes involved in the supply chain is steam cracking of naphtha and natural gas, delivering Ethylene (precursor in Styrene production), Propylene (which beyond serving as monomer for PP is used directly in Acrylonitrile production and as precursor in Styrene production), Butadiene and other relevant intermediate flows (e.g. pyrolysis gas including mixtures of Benzene, Toluene and Xylenes). Other relevant conversion processes are catalytic reforming of naphtha and steam reforming of natural gas. In catalytic reforming, naphtha is processed, to produce Benzene, Toluene and Xylenes, with Benzene being used as an intermediate for Styrene production. Steam reforming of natural gas generates synthesis gas consisting of Carbon Monoxide and Hydrogen, a precursor to Ammonia, which is used in Acrylonitrile production.

As for the steam cracking process of naphtha, the applied inventory is mainly based on industry data, completed, where necessary, by literature data. 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 co-products. 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 4.4.1.1), as the process typically takes place at refining facilities. In the subsequent separation process of reformat gas into its components (including Benzene used for Styrene 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 generate through natural gas reforming. The final polymerisation process of PP and ABS is mainly modelled based on industry data from internationally adopted production processes, integrated with literature data when needed. No allocation is performed at this stage, being PP and ABS the only outputs from the respective production process.

For fossil-based PBS, no “ready-to-use” LCI is currently available, due to the relatively early stage of development of the related production technologies. Therefore, a new LCI has been developed on purpose, using data reported by (Broeren et al. 2017). For building the new LCI, EF-compliant datasets were used where available.

#### **10.4.2.2 Recycled polymers**

The production of recycled PP granulate out of sorted, post-consumer PP waste was modelled based on based on foreground LCI data from Franklin Associates (2018), which represent mass-weighted averages of values collected from several recycling facilities in the Units States. For modelling purposes, such data were combined with background EF datasets related to energy and material supply under EU-average conditions. The overall recycling efficiency is equal to 85.5%, with removed contaminants and processes waste being sent to incineration (which along with co-combustion in cement kilns is one of the two typical fates of plastic recycling residues with high calorific value; Rigamonti et al., 2014).

According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the recycling process were allocated to the recycled content in PP panels ( $A = 0.5$  for PP used in unspecified applications). However, the recycled content carries a share of the 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 life cycle providing the recycled material). Assuming the quality of recycled PP granules is comparable to that of (panel-grade) virgin granules, a value of the  $Q_s/Q_p$  factor equal to 1 was considered. Therefore, the allocated share of virgin PP production impacts is equal to 50% ( $A \times Q_s/Q_p = 0.5 \times 1 = 0.5$ ). Virgin polymer production burdens were modelled as described above (Section 10.4.2.1) for conventional, fossil-based PP, and in Section 10.4.1.1 for the respective Feedstock Supply.

### **10.4.2.3 Bio-based polymers**

#### **10.4.2.3.1 Bio-based PBS**

The production of fully bio-based PBS out of US corn was modelled based on an aggregated dataset provided by Thinkstep. The dataset covers the steps of both succinic acid production through the enzymatic hydrolysis of corn starch followed by fermentation and filtration, and the esterification of succinic acid in the presence of butanol to produce polybutyl succinate. For more details, please refer to the dataset documentation provided by Thinkstep. It is noted that the dataset reflects European background conditions for energy and material inputs, despite the scenario assumes PBS production taking place in the US. This is acknowledged as a limitation of this study.

#### **10.4.2.3.2 PLA**

The production of PLA out of US Maize was modelled based on an aggregated, gate-to-gate dataset provided by Thinkstep. The latter covers the processes of corn wet milling for starch production, starch hydrolysis to glucose, fermentation to lactic acid, oligomerization to lactide monomer, and final polymerisation to PLA. For the corn wet milling process, economic allocation is applied to the different co-products, while for downstream conversion processes no allocation is required. The overall inventory is mainly based on industry data from internationally adopted production processes, and is completed, where necessary, by literature data. It is noted that the dataset reflects European background conditions for energy and material inputs, despite the scenario assumes PLA production taking place in the US. This is acknowledged as a limitation of this study.

### **10.4.2.4 Transport of polymer granulate to the article production site**

Modelling of transport of polymer granulate from the polymerisation plant (inside or outside the EU), to the automotive panel manufacturing site in Europe, is based on the default transport scenario (distances and vehicle types) specified in the PEF context (and in this 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 PP and ABS, and also PLA and Bio-PBS), the following routes were thus considered:

- (i) 130 km by lorry (total weight >32 t; Euro 4);
- (ii) 240 km by train (average freight); and
- (iii) 270 km by ship (barge).

For the imported share of fossil-based PP and ABS (10% and 25%, respectively), and also for PLA and Bio-PBS, a transoceanic ship transport was considered as the main transport route. The corresponding overall 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)<sup>158</sup>. Countries contributing to at least 90% of the overall import were considered in the calculation, for an overall distance equal to 8,609 km for fossil-based PP, and to 14,964 km for fossil-based ABS (see Table

<sup>158</sup> Available at: <https://www.searates.com/services/distances-time/>

10.4 and Table 10.5). For PLA and Bio-PBS (both entirely imported from the US), a distance of 6,000 km was estimated, considering the route New York – Rotterdam. Oceanic ship transport was complemented with road transport to and from the harbour in both the exporting and importing country (i.e. the EU). Road transport is made by lorry (total weight >32 t; Euro 4) along an overall default distance of 1000 km.

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

**Table 10.4.** Calculation of the overall average sea distance for imports of fossil-based PP to Europe.

Exporting country	Import <sup>(1)</sup> (%)	Import (% cum.)	Distance <sup>(2)</sup> (km)	Weighted distance (km)
SAUDI ARABIA	41.3	41.3	8767.28	3622
KOREA, REPUBLIC OF (SOUTH KOREA)	13.5	54.8	16702.31	2259
INDIA	7.15	62.0	10267.28	734
RUSSIAN FEDERATION (RUSSIA)	6.75	68.7	5574.44	376
BRAZIL	6.63	75.4	10107.97	670
ISRAEL (GAZA and JERICHO- >1994)	4.86	80.2	2987.48	145
SOUTH AFRICA (incl. NA - >1989)	4.26	84.5	12160.92	518
EGYPT	3.20	87.7	3212.14	103
UNITED STATES	2.99	90.7	6061.90	181
Other countries	9.32	100	-	-
<i>Overall weighted distance</i>				<i>8,609</i>

<sup>(1)</sup> Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

<sup>(2)</sup> 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 Middle-East and Asia were determined considering Marseille as destination port in Europe. For imports from other countries (in the case of PP Russia, Brazil, South Africa and United States), Rotterdam was considered as destination port.

**Table 10.5.** Calculation of the overall average sea distance for imports of fossil-based ABS to Europe.

Exporting country	Import <sup>(1)</sup> (%)	Import (% cum.)	Distance <sup>(2)</sup> (km)	Weighted distance (km)
KOREA, REPUBLIC OF (SOUTH KOREA)	70.1	70.1	16702.31	11714
TAIWAN	16.8	87.0	15336.64	2583
THAILAND	4.90	91.9	13618.69	667
Other countries	8.13	100.00	-	-
Overall weighted distance				14,964

<sup>(1)</sup> Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

<sup>(2)</sup> 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 Middle-East and Asia were determined considering Marseille as destination port in Europe. For imports from other countries (none in the case of ABS), Rotterdam was considered as destination port.

### 10.4.3 Article Production Stage

Regardless of the feedstock and polymer used, manufacturing of automotive interior panels normally takes place via injection moulding of melted plastic granules. In this process, melted or plasticized plastic material is injected by force into a clamped mould cavity (consisting of a single cavity or a number of similar or dissimilar cavities), where it is finally shaped into the desired object (Rosato et al., 2004).

The burdens of the injection moulding process were modelled through the aggregated, EF-compliant dataset Injection moulding, plastic injection moulding, production mix, at plant [EU-28+EFTA], which accounts for nearly 3% polymer losses during the process. The burdens of the respective disposal process were additionally included in the model, assuming that they are entirely sent to recycling or incineration (depending on the polymer).

For PP (from all feedstock sources) and ABS, process losses 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 PP and ABS polymer granules were assumed to replace virgin, fossil-based granules of the same polymer. The recycling process and resulting virgin material substitution was modelled based on the same data as End of Life recycling of sorted, post-consumer PP and ABS panels, in the absence of more specific data for recycling of pre-consumer, industrial scraps. For further detail on the modelling of recycling and avoided virgin material production (including the implementation of the Circular Footprint Formula), the reader is hence referred to Section 10.4.5.2 on End of Life modelling.

For non-conventional polymers, i.e. PBS, Bio-PBS and PLA, process losses were assumed to be incinerated (see section 10.4.5.3).

### 10.4.4 Distribution Stage

The transport of automotive panels from the manufacturing site to the final user was modelled based on the default transport scenario specified in the PEF context (and in the Method) for the route *factory* → *retail* → *final client*. The following routes were thus considered:

- i) 1200 km by lorry (total weight >32 t; Euro 4) from factory to retailers;
- ii) 5 km by van for 5% of the roundtrips from retailers to final users; and

- iii) no burdens assigned to 33% of the roundtrips from retailers to final users  
(assumed to take place with no motorised vehicles).

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

#### **10.4.5 End of Life Stage**

This section describes the modelling of the End of Life stage of automotive interior panels. In particular, Section 10.4.5.1 focuses on the definition of the EU-average End of Life scenario, which is considered as a base case for the calculation of the potential impacts of the compared LCA scenarios. The remaining sections (10.4.5.2 – 10.4.5.6) address the modelling of the different End of Life options explored in the study. Finally, Section 10.4.5.7 provides case-specific details on the estimate of the potential contribution of automotive panels to macro-plastic formation at the End of Life (including product litter) and micro-plastic generation throughout the supply chain.

##### **10.4.5.1 End of Life scenario**

Regardless of the type of feedstock, the average End of Life scenario of automotive interior panels was assumed to include 56% recycling after separate collection, 41% reuse, 2% incineration and 1% landfilling. These rates are based on average EU-28 data of vehicles End of Life (large plastic components) as reported by Eurostat (2018). The collection of panels and its transport to the corresponding waste management option was not considered in this case study due to lack of data.

In order to evaluate the burdens of each automotive panel scenario when individually applying the different viable End of Life options, namely recycling, incineration, landfilling, and (for panels made of biodegradable polymers) composting and anaerobic digestion (AD), additional scenarios were modelled considering 100% of waste panels sent to either one of the aforementioned End of Life options. Although not common practice for automotive interior panels, 100% AD and 100% composting were considered here, in order to illustrate how they could affect the environmental performance of the plastic article over its life cycle.

##### **10.4.5.2 Modelling of recycling**

No specific dataset is available in the EF context for PP recycling. Hence, a new dataset was developed based on foreground inventory data available in Franklin Associates (2018), combined with EF background datasets for energy and material supply under EU-average conditions. The inventory is based on mass-weighted average values of data collected from several recycling facilities in the United States. While these data are mostly representative of recycling of PP products from municipal solid waste collection, they were considered suitable to also approximate recycling of dismantled PP panels, which is expected to rely on similar unit operations (e.g. grinding/shredding, washing/flotation and granulation). The overall recycling efficiency is equal to 85.5%, with removed contaminants being sent to incineration along with process waste. This is in line with the typical fate of recycling residues, which due to their high calorific value are normally sent to incineration or co-combustion in cement kilns (Rigamonti et al., 2014).

Recycling of ABS, PBS, Bio-PBS and PLA was modelled using the EF dataset *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*, which was applied in the absence of more specific and representative data for recycling of the mentioned polymers. 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). Process waste and scrap are sent to incineration, consistently with the typical fate of plastic recycling residues, and the assumption performed in the PP recycling process.

Recycled polymer granulate was assumed to replace virgin granulate of the same material, whose primary production burdens were credited to the system. To account for the lower overall quality of recycled polymers compared to the replaced virgin material, a substitution ratio equal to 0.9 was considered, according to the default values specified in the PEF context for thermoplastic polymers (e.g. PET, PE and PP) and in the absence of specific data for some of the investigated polymers. To model the burdens of avoided primary production, the same datasets (or combination of datasets) used for the modelling of upstream production of the relevant polymers were applied (as described in Sections 10.4.1 and 10.4.2 for the related feedstock). This was made for consistency reasons, and avoiding possible distortions by applying different datasets from other sources. According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the recycling process were allocated to the system ( $A = 0.5$  for PP used in unspecified applications, and extending the same assumption to the remaining polymers). Similarly, only 50% of the benefits from avoided virgin material production were assigned to the system itself.

#### **10.4.5.3 Modelling of incineration**

For PP, ABS and PLA, aggregated, material-specific incineration datasets are available in the EF or GaBi database, and were applied to model the treatment of automotive panels made of these materials in a municipal waste incineration plant. All the selected datasets are based on a waste-specific incineration model, which has been described more in detail in Section 4.4.5.4. 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 general approach to handle energy recovery situations adopted in the EF context and in the Method (Report I), the product system generating the waste material sent to incineration (i.e. the automotive interior panels life cycle, in this case) takes the full burdens from the incineration process. However, it is credited for 100% of the benefits from avoided production of conventional energy (electricity and heat) replaced by energy recovered from waste. While in EF incineration datasets these credits are already accounted for in the aggregated inventory, for GaBi datasets they were added to the main process inventory. Therefore, the EU residual electricity grid mix (as modelled in the EF 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 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 includes 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. Also, thermal energy from waste (11%) was excluded, as according to the approach described above for the handling of energy recovery situations, 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 PBS and bio-PBS, no ready-to-use incineration datasets were available. A disaggregated, material-specific inventory was thus developed, based on the most recent version of the calculation tool developed by Doka (2009a) for the modelling of material incineration within municipal solid waste incineration plants. The model operates similarly to the one used for the development of EF incineration datasets, allowing the practitioner to account for the specific composition and energy content of the incinerated waste (see

Table 10.6 for an overview of the considered values) to develop a material-specific incineration inventory based on transfer coefficients. The tool also allows to adjust other technological parameters to the relevant geography or scope, including energy efficiencies, the share of alternative NO<sub>x</sub> control technologies, and a few other specific parameters. In this case 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, and applied in the modelling. These efficiencies account for the share of waste routed to incineration plants operating without any energy recovery (estimated to be 9% for municipal waste)<sup>159</sup>, while considering that plants with energy recovery operates with an average gross electricity efficiency equal to 15.1%, and a gross thermal efficiency equal to 35% (CEWEP, 2012)<sup>160</sup>. Credits associated with recovered energy were calculated as described above for existing GaBi datasets (i.e. electricity from the EU-average 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, background EF datasets were applied.

#### 10.4.5.4 Modelling of landfilling

Landfilling of automotive panels made of non-biodegradable polymers (i.e. fossil-based PP and ABS, both virgin and recycled) was modelled based on a common, aggregated, EF dataset representing disposal of plastic waste in a managed, municipal solid waste landfill ([EU-28+EFTA] Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific sites)). The underlying inventory is material-specific, but refers to the average chemical composition and degradability values of generic plastic waste, rather than to those of the specific polymer. This is considered an acceptable approximation for the scope of this study, since degradation rates in the landfill body (one of the most relevant parameters for landfill modelling) is similar for all non-biodegradable (conventional) polymers<sup>161</sup>. 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 emissions to the environment over a 100 year time horizon. Emissions occurring beyond 100 years are not accounted for in the model. Landfill gas generation is calculated based on the organic carbon content in the waste and the respective degradation rate assumed over 100 years (not reported), while, 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 rate of landfill gas, energy efficiencies of engines, collection rate of leachate and respective treatment efficiencies). Further details on how these parameters were set to reflect the EU-average situation are provided in Section 4.4.5.5.

For automotive panels made of non-“drop-in” biodegradable polymers (i.e. PBS, Bio-PBS and PLA), a material-specific landfilling inventory was built, based on the calculation tool developed by Doka (2009b) for the modelling of waste disposal into sanitary landfills. The tool allows to calculate material-specific landfilling inventories accounting for the specific chemical composition and other relevant chemo-physical properties of the

<sup>159</sup> Calculated as average share of municipal waste incinerated without energy recovery over the years 2015-2017 (Eurostat, 2019d).

<sup>160</sup> 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).

<sup>161</sup> While more polymer-specific landfilling datasets are available from other databases (i.e. *ecoinvent*), the mentioned EF dataset for landfilling of generic plastic was selected, as specifically referring to EU as the reference geography (in contrast to available polymer-specific datasets), and to keep consistency with the overall dataset selection “hierarchy” followed in this project (Section 3.5).



landfilled waste (Table 10.6), as well to adjust a number of relevant site- and technology-specific parameters to the reference geography and to the corresponding average landfilling technology. For this purpose, the model applies element-specific transfer coefficients to calculate the distribution of decomposition products originating from elements in the waste composition between landfill gas and leachate, and their ultimate emissions 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.

**Table 10.6.** Elemental composition and lower heating value of PBS/Bio-PBS and PLA considered for the End of Life modelling.

Element	(%)	Element	(%)
<b>PBS/Bio-PBS <sup>(1)</sup></b>			
TS	99.9	C (%TS)	55.8
Water	0.1	H (%TS)	7.0
VS (%TS)	100	O (%TS)	37.2
Ash (%TS)	0	C <sub>biogenic</sub> (% total C)	0/100
LHV = 22.1 MJ/kg TS <sup>(2)</sup>			
<b>PLA <sup>(3)</sup></b>			
TS	100	H (%TS)	5.60
Water	0	O (%TS)	44.5
VS (%TS)	100	N (%TS)	0.1
Ash (%TS)	0	S (%TS)	0.3
C (%TS)	49.5	C <sub>biogenic</sub> (% total C)	100
LHV = 18.4 MJ/kg TS <sup>(2)</sup>			

<sup>(1)</sup> Based on stoichiometry.

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

<sup>(3)</sup> Based on results from composition analysis related to Ingeo polymer available at:  
<https://www.natureworksllc.com/What-is-Ingeo/Where-it-Goes/Incineration>

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. carbon to CH<sub>4</sub> and CO<sub>2</sub>) within the landfill. For PLA panel, a 1% biodegradation was assumed, according to the results from Accelerated Landfill Conditions (ALC) tests simulating approximately 100 years of deposition in a biologically active landfill (390 days at 21 °C, Kolstad et al., 2012). These showed no significant biodegradation of both amorphous and semi-crystalline PLA over such a time horizon, even at the highest humidity level applied for testing (i.e. 65%, which should be considered in a conservative approach). Therefore, the same value of degradability assumed in the model for conventional, non-biodegradable polymers such as PET and PE was consistently applied (i.e. 1%). In the absence of Accelerated Landfill Conditions (ALC) tests for PBS, 1% biodegradation was assumed for PBS and Bio-PBS

polymers, in light of the negligible biodegradation of these materials under anaerobic conditions (see section 10.4.5.6).

In line with the time horizon applied for landfill emission modelling in this tool and in earlier reported datasets, (biogenic) carbon in the landfilled polymers that is not degraded (mineralised) after 100 years from deposition, was considered to be never released from the landfill body. As a base case, this missed release of atmospheric carbon taken up during biomass growth is not reflected in Climate Change LCIA results for PLA and Bio-PBS automotive panels, since characterisation factors for biogenic CO<sub>2</sub> flows have been set to zero (fully conforming to provisions in the PEF method). However, to show the relevance of this choice on the overall results, the Climate Change impact of the two bio-based alternatives was also calculated by acknowledging the effects of non-released biogenic carbon. For this purpose, a specific CO<sub>2</sub> uptake was modelled in the inventory, expressing the net amount of CO<sub>2</sub> taken up during biomass growth and not released (as CO<sub>2</sub> or CH<sub>4</sub>) during polymer degradation in landfill. The uptake was calculated based on the biogenic carbon content in the polymer (Table 10.6), and assuming a mineralisation rate over 100 years equal to 1% for both Bio-PBS and PLA (as discussed above). A carbon uptake equal to 1.80 kg CO<sub>2</sub>/kg PLA and 2.03 kg CO<sub>2</sub>/kg bio-PBS was thus calculated. This uptake was then characterised with a factor of -1 kg CO<sub>2</sub> eq. per kg CO<sub>2</sub> not released.

Site-specific and technology-specific parameters were set so as to reflect as much as possible the average situation at the EU level, as detailed in Section 4.4.5.5. These include the rates of landfill gas capture, utilisation and flaring, as well as the mean annual precipitation and temperature. For the other parameters (e.g. energy efficiencies of gas engines, mean annual evapotranspiration, landfill height and duration of the filling phase) the default values assumed in the model were kept.

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 database<sup>162</sup>. However for energy-related flows, background EF datasets were applied.

#### **10.4.5.5 Modelling of composting and compost use-on-land**

A waste-specific life cycle inventory was developed for aerobic composting of biodegradable automotive panels (i.e. PBS, Bio-PBS and PLA), following the approach and the assumptions described in Section 5.4.5.6 (to which the reader is referred for further details beyond those specified below). The modelling was carried out according to the general recommendations specified in the Method (Report I - Sections 4.4.12.4 and 4.4.12.7), while relying on process-specific burdens and element-specific transfer coefficients available in the EASETECH model (Clavreul et al., 2014) for the chosen composting technology (i.e. tunnel composting, as discussed below).

The elemental composition assumed for modelling purposes for each panel material is the one already reported in Table 10.6, while the applied biodegradation rates are summarised in Table 10.7. The latter were defined as the average of (experimental) values available in the literature for fragments of plastic sheets/bars with different thicknesses. Since these values were at least partly determined under real composting or laboratory conditions, they were preferred to the application of the minimum biodegradation rate required by the European standard to evaluate compostability of plastics (EN 14995) i.e. 90%. This has indeed to be proven under optimised laboratory testing conditions and may not be achieved in real composting plants (or it may anyway differ from values obtained from real laboratory testing available in the literature).

Compared to packaging articles investigated in former case studies (i.e. food packaging film and food trays), automotive panels were assumed to be firstly grinded into smaller pieces (e.g. together with green waste possibly received by the plant) to allow proper

<sup>162</sup> Exceptions are the inputs of diesel (burned in building machine) and pitch, which were replaced with suitable EF datasets, to improve reliability of LCIA results in the Ozone Depletion impact category.

degradation (larger articles like panels are likely unsuitable for direct degradation within composting facilities). Moreover, consistently with the composting process already modelled for the mentioned packaging articles a further screening step was considered, where 30% of the incoming (grinded) material is removed and sent to incineration (modelled as described in Section 10.4.5.3).

Carbon in biodegradable automotive panels sent to composting was assumed to mineralise according to the biodegradation rates reported in Table 10.7. The same rates were also assumed for biodegradation of volatile solids (VS) in each panel materials. According to the emission factors (transfer coefficients) reported in the EASETECH model, 99.8% of mineralised carbon is converted to CO<sub>2</sub>, the rest (0.2%) being converted to CH<sub>4</sub>. The latter is then mostly oxidised to CO<sub>2</sub> (95%), while only 5% is eventually emitted as Methane, equalling 0.01% of the mineralised carbon. No waste-specific Nitrogen emissions from the composting process were modelled, being the Nitrogen content of the investigated biodegradable polymers equal to zero or negligible (e.g. for PLA).

The amount of residual composted material (biomass and simpler organic compounds) obtained from the biodegradation of the automotive panels was calculated as the sum of non-degraded volatile solids, as well as of ash and water<sup>163</sup> originally included in the material sent to composting. The residual material was assumed to be entirely applied on agricultural land, where biodegradation of the residual material further advance. Application is made by means of tractors, with the specific diesel consumption equal to 0.00042 l/kg composted material applied. The emissions from on-land application were modelled based on average emission factors estimated by Bruun et al. (2006), considering a time horizon of 100 years from application. Considering average factors across all types of temperate soils, 89.3% of the applied carbon was assumed to mineralise to Carbon Dioxide, 0.01% was released as Methane, while the remaining 10.7% was not released<sup>164</sup> (being part of more stable organic compounds).

The overall composting inventory was implemented in the lifecycle model according to the Circular Footprint Formula, considering it as a case of recycling with a value of the A factor equal to 0.5. Only 50% of the burdens from the composting process were thus allocated to the automotive panel life cycle. Conversely, the burdens and benefits from on-land application of the residual composted material were entirely allocated to it.

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<sup>163</sup> It is acknowledged that the water content in the residual composted material may change compared to the original material sent to composting due to losses and integration during the process. However, the water content of the investigated biodegradable polymers is equal to zero, so that this simplifying assumption does not affect the calculated amount of residual material from composting.

<sup>164</sup> As a base case, this non-emission of biogenic carbon is not reflected in Climate Change LCIA results of (Bio)-PBS and PLA automotive panels, since characterisation factors for biogenic CO<sub>2</sub> flows have been set to zero (fully conforming to provisions in the PEF method). The consequences of this are not significant, since the share of residual, non-emitted carbon is low, equalling 0.5-5% of total carbon in the composted polymers. However, to show the relevance of this missed quantification on comparative scenario results, the Climate Change impact of PBS/Bio-PBS and PLA automotive panels was also calculated by acknowledging the effects of non-released biogenic carbon. For this purpose, a specific CO<sub>2</sub> uptake was modelled in the composting inventory, corresponding to the amount of non-released biogenic carbon, which is equal to 0.0654 kg CO<sub>2</sub>/kg PLA panels sent to composting, and 0.00764 kg CO<sub>2</sub>/kg PBS/Bio-PBS panels.

**Table 10.7.** Biodegradability values considered for biodegradable automotive interior panels (PLA, PBS and Bio-PBS) under aerobic composting conditions.

Polymer	Biodegradation rate (%)	Source
PLA	52 <sup>(1)</sup>	OWS (2013) – Grinded bars; 3.2 mm thickness (88%) Hermann et al. (2011) – Film fragments; 1 mm thickness; 1 cm <sup>2</sup> area (55%) UBA (2018) – Pot fragments (13%)
PBS/Bio-PBS	95 <sup>(2)</sup>	UBA (2018) Deconinck and De Wilde (2013)

<sup>(1)</sup> Calculated as the average of (experimental) results for fragments of PLA sheets/bars with 1 mm and 3.2 mm thickness and pot fragments.

<sup>(2)</sup> Calculated as the average of experimental results for Bionelle polymer.

#### 10.4.5.6 Modelling of anaerobic digestion

Anaerobic digestion of PLA panels and subsequent post-composting of the resulting, non-digested or partially digested, residual material were modelled according to the approach and the assumptions described in Section 5.4.5.7 (to which the reader is referred to for further details beyond those specified below). For anaerobic digestion, a waste-specific inventory was developed, based on the general recommendations specified in the Method (Report I – Section 4.4.12.5), while relying on relevant process-specific burdens and element-specific transfer coefficients available in the EASETECH model (Clavreul et al., 2014) for the chosen digestion technology (i.e. wet thermophilic digestion).

The elemental composition considered in the modelling for automotive panels is the one already reported in Table 10.6, while the applied biodegradation/mineralisation rate equal to 25% was applied, according to the lowest value reported for PLA film in the extensive review conducted by UBA (2018; no rates are available for thicker PLA films/sheets). Since this value refers to a film with a thickness of 0.1 mm, which is lower than the one assumed for automotive panels (0.8 mm), the modelled digestion scenario is optimistic in terms of applied degradation rate. However, being such a value derived from real testing (albeit it is not specified whether under laboratory or real plant conditions), it was preferred to the application of the minimum biodegradation rate required by the European standard on compostability of plastics (EN 14995), i.e. 50% when testing anaerobic treatability.

Similarly to the process modelled for packaging articles, prior to digestion discarded panels were assumed to undergo a pre-treatment stage, where they are also shredded to make them suitable for further processing in the digestion plant. In this stage, 30% of automotive panels collected for treatment was assumed to be removed as residue, and sent to incineration (modelled as described in Section 10.4.5.3).

Carbon in PLA panels sent to digestion was assumed to be converted (mineralised) into biogas according to the 25% biodegradation rate reported above. The shares of gasified carbon converted into Methane and Carbon Dioxide were calculated based on the stoichiometry of the anaerobic degradation reaction, taking into account the proportions of C, H, O and N in the digested polymer. Carbon conversion shares are thus material-specific and, in the case of PLA, they are equal to 50% for both Methane and Carbon Dioxide.

The amount of residual, non-digested or partially digested bioplastic material in the digestate output was calculated as the sum of non-degraded volatile solids (VS), as well as of ash and water originally present in the input material to digestion. Residual volatile solids were in turn calculated by assuming a ratio between degraded VS and degraded carbon equal to 1.89 (according to the value applied in the EASETECH model for organic waste).

Aerobic post-composting of such residual material from digestion was modelled considering an open (windrow) composting facility, and the typical average carbon (and VS) biodegradation (mineralisation) rate achieved for generic organic waste in this type of facilities (i.e. 76.4% according to the considered EASETECH dataset). Based on transfer coefficients (emission factors) reported in the latter, 97.76% of mineralised carbon was assumed to be converted to CO<sub>2</sub>, and the remaining 2.24% to CH<sub>4</sub>.

The residual organic material from post-composting of digested panels was assumed to be applied on agricultural land as soil amendment. On-land application was modelled according to the same approach and assumptions briefly described in Section 10.4.5.5 for residual material derived from direct composting of PLA panels (and more extensively detailed in Section 5.4.5.6 for the food packaging film case study).

#### **10.4.5.7 Contribution to macro- and micro-plastics generation (including product litter)**

The contribution to macro- and micro-plastics loss and release of the analysed automotive interior panels scenarios was estimated according to the framework outlined in Annex B of the Method (Report I), and summarised in Section 4.4.5.6 of this report in terms of underlying quantification methods and approaches. In this section, the case-specific parameters considered to apply the latter to the assessed automotive interior panels LCA scenarios are reported.

##### **10.4.5.7.1 PLP method**

Regarding the estimate of the loss and release of macro-plastics at the End of Life stage (from product littering and waste mismanagement), the product-specific parameters reported in Table 10.8 were considered to apply the first level of the framework (“PLP method”) to automotive interior panels (derived from Peano et al., 2020). Note that such parameters are defined regardless of the type of feedstock or material used for panels manufacturing. Similarly, all automotive interior panels scenarios contribute to macro-plastics loss and release to the same extent, as the chosen feedstock or material do not affect the probability of the article to be littered or mismanaged.

**Table 10.8.** Case-specific parameters of the PLP method applied to the automotive interior panels LCA scenarios.

<b>Parameter <sup>(1)</sup></b>	<b>Value</b>
Littering rate (LR <sub>lit</sub> ) (%)	0
Release rate to ocean (Rel <sub>ocean</sub> ) (%)	5
Release rate to the terrestrial environment (Rel <sub>terenv</sub> ) (%)	95

<sup>(1)</sup> For details on the meaning of each parameter, refer to Annex B of the Method (Report I).

No case-specific parameters are required to calculate the supply-chain loss and release of micro-plastics to ocean and to the terrestrial environment by applying the PLP method. Indeed, the contribution of the considered micro-plastics sources (pellet loss and tire abrasion) depends on the magnitude of the relevant life cycle processes (i.e. polymer production and road transport) in the specific automotive interior panels scenario (which in turns depend on the reference flow of each scenario). However, the quantification is made by means of default parameters that are not affected by the type of product, polymer or feedstock source (as reported in Annex B of the Method (Report I)).

##### **10.4.5.7.2 Expanded PLP method**

The parameters required to calculate the micro-plastics loss and release from marine coatings and road markings are not case-specific, and are reported in Annex B of the Method (Report I). Similar to the other considered micro-plastics sources, the contribution of these additional sources depends on the intensity of relevant (transport)

processes in the specific product inventory, by means of the above-mentioned case-  
unspecific parameters.

As for the calculation of the additional contribution of plastic waste exported for recycling  
to the loss and release of macro-plastics from waste mismanagement, the case-  
unspecific parameters described in Annex B of the Method (Report I) were applied, in  
combination with the EU-average recycling rates estimated for each type of automotive  
interior panels (Section 10.4.5.1), i.e. 56% for all scenarios.

#### 10.4.5.7.3 Sensitivity analysis 1: alternative calculation of the total plastic loss along the supply-chain

The contribution of each (foreground) life cycle process to the total plastic loss of the  
different automotive interior panels scenarios was quantified by applying the approach by  
Ciroth & Kouame (2019) to the corresponding *ecoinvent* dataset or, if needed, adapted  
dataset from the same database<sup>165</sup>. Table 10.9 summarises the datasets considered for  
the cradle-to-gate process-chain involved in the production of each polymer, along with  
the possible adaptations performed. The datasets considered for the remaining  
foreground or background lifecycle processes are instead listed in Annex D. Note that this  
approach excludes the contribution from product littering and waste mismanagement at  
End of Life.

**Table 10.9.** Ecoinvent datasets applied and possible adaptations performed to quantify the total  
plastic loss from the production of the different polymers considered for automotive panel  
manufacturing, through the approach developed by Ciroth and Kouame (2019).

Polymer	Dataset	Adaptation
PP	[RER] Polypropylene terephthalate production, granulate, bottle grade	No adaptation required.
ABS	[RER] Acrylonitrile butadiene styrene, granulate	Adaption with addition of feedstock to the system process dataset.
PLA	[RER] Polylactide production, granulate	No adaptation required.
Recycled PP	Generic recycled plastic dataset developed on purpose.	No adaptation required.
PBS	[RER] Succinic acid production	No adaptation required.
Bio-PBS	[RER] Succinic acid production	Substitution of maleic acid by maize grain from global market.

#### 10.4.5.7.4 Sensitivity analysis 2: alternative calculation of macroplastics release to ocean

This sensitivity analysis was not applied to automotive interior panels scenarios due to  
the absence of automotive interior panels in the EU beach litter database (Addamo et al.,  
2018; Hanke et al., 2020).

<sup>165</sup> Equivalent to those applied in the actual lifecycle model of each scenario, which could not be directly  
considered for calculation purposes as the mentioned approach has been specifically developed to be  
applied to *ecoinvent* inventories.

#### **10.4.6 Calculation of iLUC impacts**

As a base case, the contribution of Indirect Land Use Change (iLUC) to the potential Climate Change impact of bio-based automotive panel alternatives was calculated according to the approach outlined in Section 4.4.16.4 of the Method (Report I).

In order to apply (recalculated) iLUC GHG emission factors from the EU 2015/1513 Directive (EC, 2015), the specific land demand for the production of relevant crops ( $\text{m}^2 \cdot \text{year} / \text{kg crop}$ ) was calculated first. The calculation was based on the aggregated amount of arable and agriculture land occupation flows reported, for the relevant country (e.g. Brazil for sugarcane), in the dataset used to model the production of the specific crop. If the geography of such flows was not specified, all the arable and agricultural land occupation flows reported in the dataset were aggregated. These estimates were checked against values of land demand calculated based on 5-years average crop yields from FAOSTAT (FAO, 2019), which were found to be generally aligned with the former (absolute variation between 3% and 20%). Hence, the values estimated based on land occupation flows were applied, for consistency with the overall LCI modelling of scenarios.

Specific land demand for crop production was then converted into a demand per functional unit, based on the specific consumption of crop for polymer production ( $\text{kg crop/kg polymer}$ , consistently with the values applied in the LCI modelling) and the amount of polymer needed to fulfil the functional unit (reference flow) in the relevant scenario. The iLUC contribution to the Climate Change impact was finally calculated by applying to the latter the recalculated GHG emission factors from the Directive. The described calculation steps to estimate the iLUC contribution to the potential Climate Change impact are summarised in Table 10.10.

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**Table 10.10.** Calculation of the iLUC contribution to the potential Climate Change impact of automotive interior panel LCA scenarios relying on bio-based polymers.

Scenario / Polymer	Feedstock	Land demand for crop production <sup>(1)</sup> [m <sup>2</sup> *y/kg <sub>crop</sub> ]	Crop demand for polymer production [kg <sub>crop</sub> /kg <sub>polymer</sub> ]	Polymer demand per F.U. [kg <sub>polymer</sub> /FU]	iLUC GHG emission factor [kg CO <sub>2</sub> eq./ (m <sup>2</sup> *y)]	iLUC Climate Change impact [kg CO <sub>2</sub> eq./FU]
S5 – PLA	Maize (US)	1.06 (0.939)	3.08	1.02	0.061	0.204
S6 – Bio-PBS	Maize (US)	1.06 (0.939)	2.78	1.02	0.061	0.184

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(1) Based on arable and agriculture land occupation exchanges reported in the dataset applied to the modelling of the specific crop. Values in parenthesis refer to land demand calculated based on crop yield data from FAOSTAT (5-years average), and are reported as a reference.



## 10.5 Life Cycle Impact Assessment results

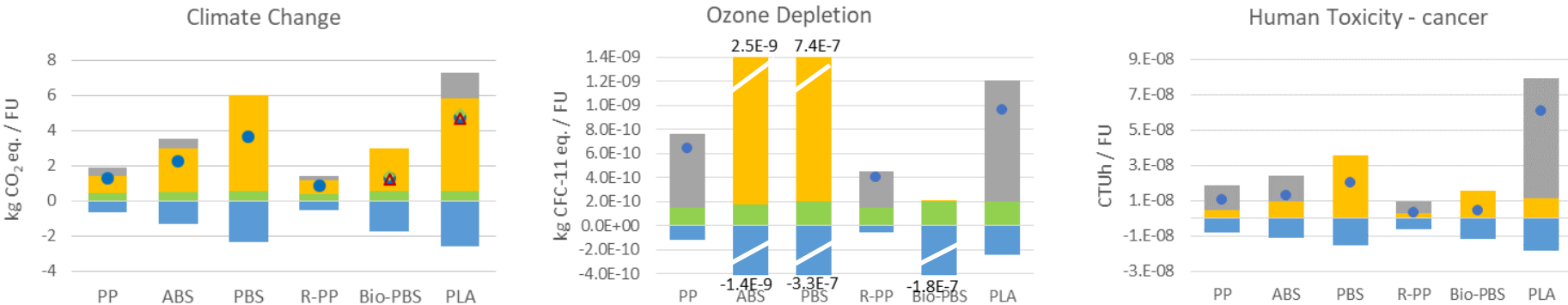
The characterised potential impacts of the examined scenarios (with and without the iLUC contribution to Climate Change) are reported in Figure 10.7 to Figure 10.9. These also show the breakdown of contributions from the main lifecycle stages, which include:

- i) *Feedstock Supply*, i.e. oil/natural gas extraction, transport and possible refining, transport of naphtha from refinery to downstream users; collection, transport and sorting of post-consumer plastic waste (recycled polymers); or crop cultivation and transport to further processing (bio-based polymers);
- ii) *Polymer Production*, i.e. all gate-to-gate activities from feedstock processing to monomer/intermediate production and polymerisation, including any transport among these, as well as transport of polymer granulate to the automotive panel manufacturing site;
- iii) *Article Production*, i.e. conversion of the polymer into automotive interior panels by injection moulding;
- iv) *Distribution*, i.e. transport of automotive panels from the manufacturing site to the final user; and
- v) *End of Life*, i.e. waste panel treatment or disposal, as well as any avoided processes from downstream displacement of virgin materials and energy.

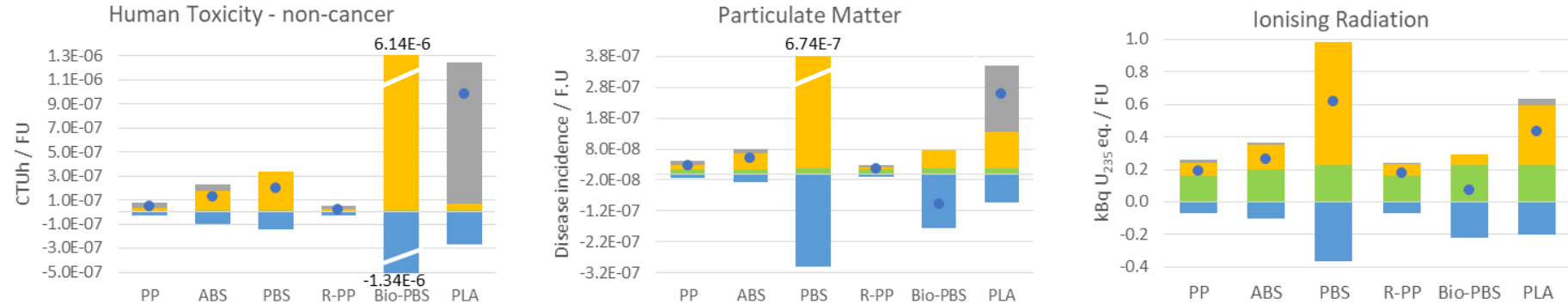
Figure 10.10 shows the contribution of each sub-category to the Climate Change impact (i.e. from fossil, biogenic and land use/land transformation GHG emissions), while the overall characterised, normalised and weighted impact assessment results are available in Annex B.7. The latter are not reported in this section to keep readability of the document, despite this is not in alignment with the position suggested in the LCA report template provided in the underlying Method (Report I).

Note that scenario impacts presented in Figure 10.7 to Figure 10.9 refer to the EU-average End of Life scenario (as described in Section 10.4.5.1), and represent the net impacts from waste management activities, resulting from the balance between real burdens and benefits (if any). Potential impacts calculated assuming 100% of post-consumer panels being routed to each viable End of Life option are presented in Figure 10.11 to Figure 10.13. We warn that this analysis should not be interpreted as a direct LCA comparison of alternative End of Life management options for automotive panels, because a product-based perspective is considered, and allocations among consecutive product life cycles are applied to the recycling pathway. Conversely, in a typical waste management LCA that compares alternative End of Life options (i.e. with a functional unit of 1 tonne of waste managed), the recycling pathway would be assigned the full burdens and benefits it involves, as there is no need to break the system mass flows between the system recycling and the one using the recycled feedstock (i.e. system perspective; no allocation needed).

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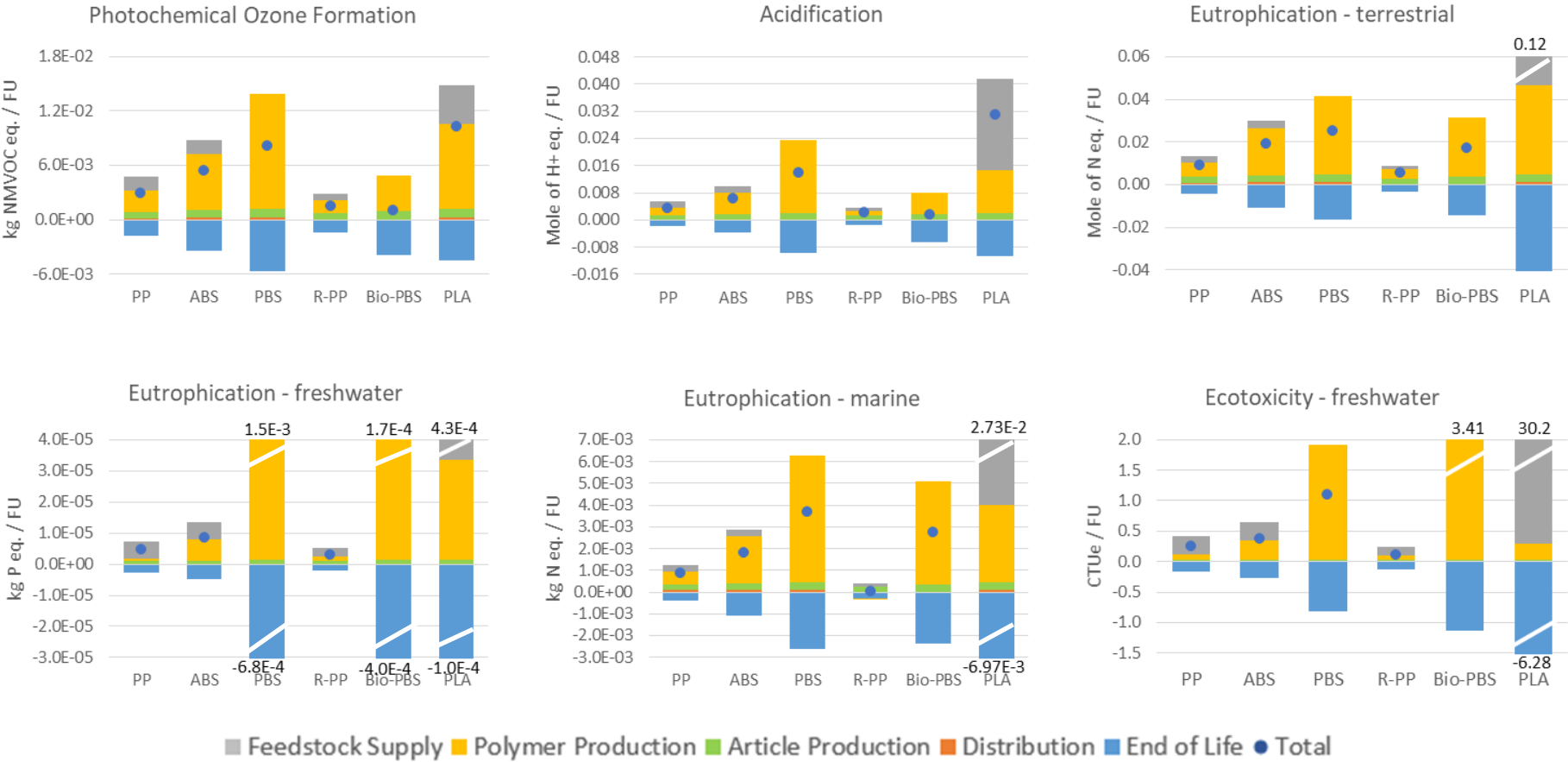
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■ Feedstock Supply ■ Polymer Production ■ Article Production ■ Distribution ■ End of Life ● Total ▲ Total w. C bio EoL ◆ Total + iLUC

4 **Figure 10.7.** Potential impact of automotive interior panels LCA scenarios for the categories of Climate Change, Ozone Depletion, Human Toxicity -  
5 cancer, Human Toxicity – non-cancer, Particulate Matter and Ionising Radiation. Note that in some impact categories a part of the results is out of scale  
6 and is curtailed. Climate Change impact denoted with “C bio EoL” accounts for the contribution of biogenic carbon not released after 100 years from  
7 landfilling of bio-based automotive panels or from on-land application of residual organic matter derived from their composting or anaerobic digestion.



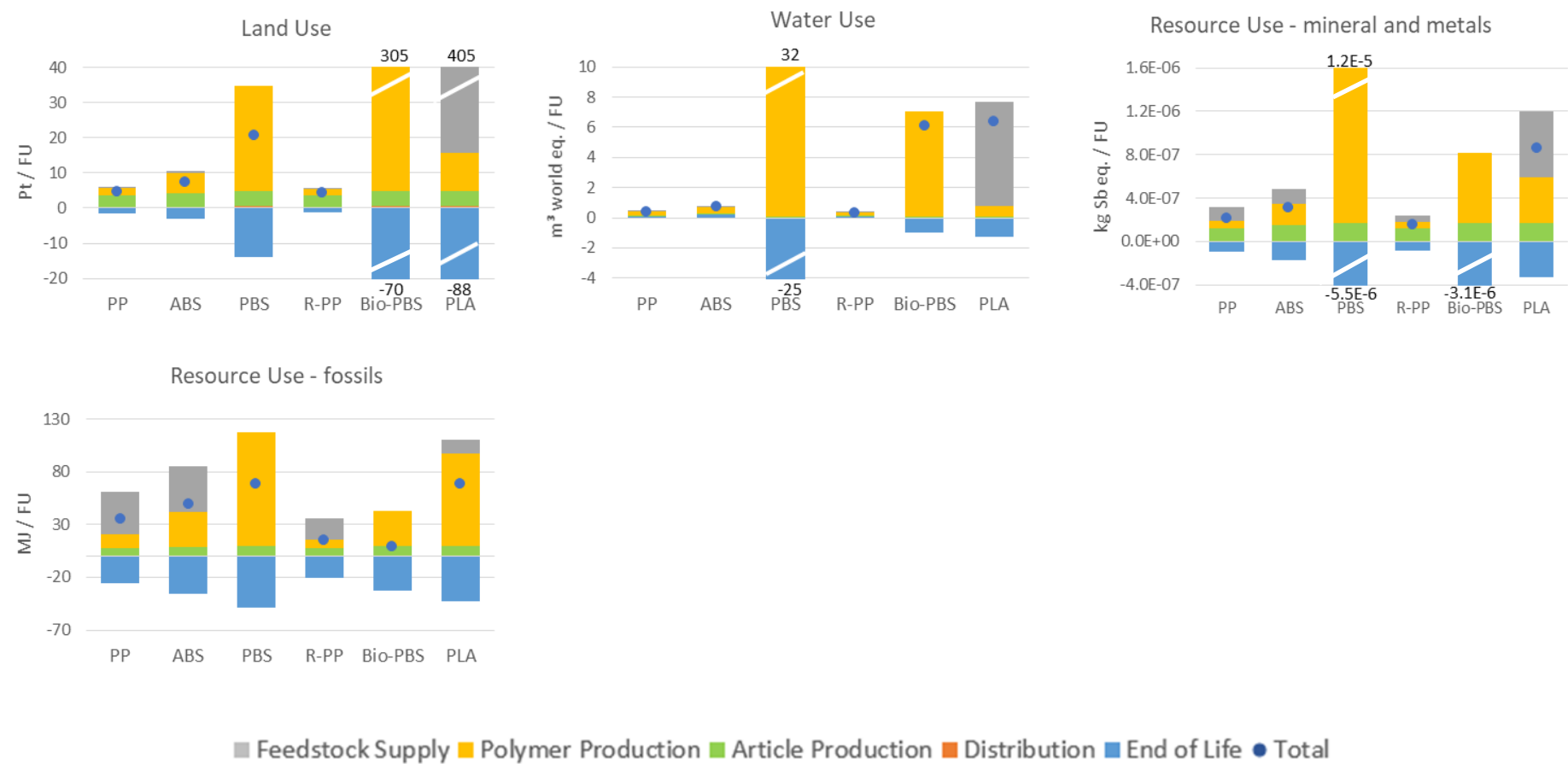
**Figure 10.8.** Potential impact of automotive interior panels LCA scenarios for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity - freshwater. Note that in some impact categories a part of the results is out of scale and is curtailed.

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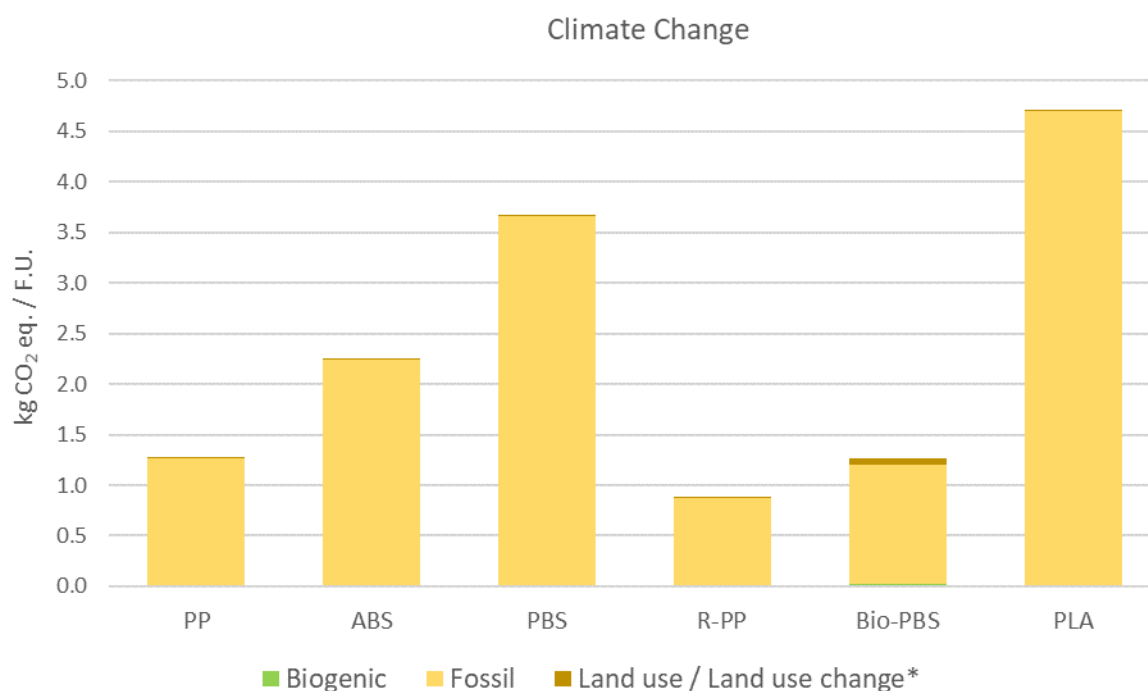
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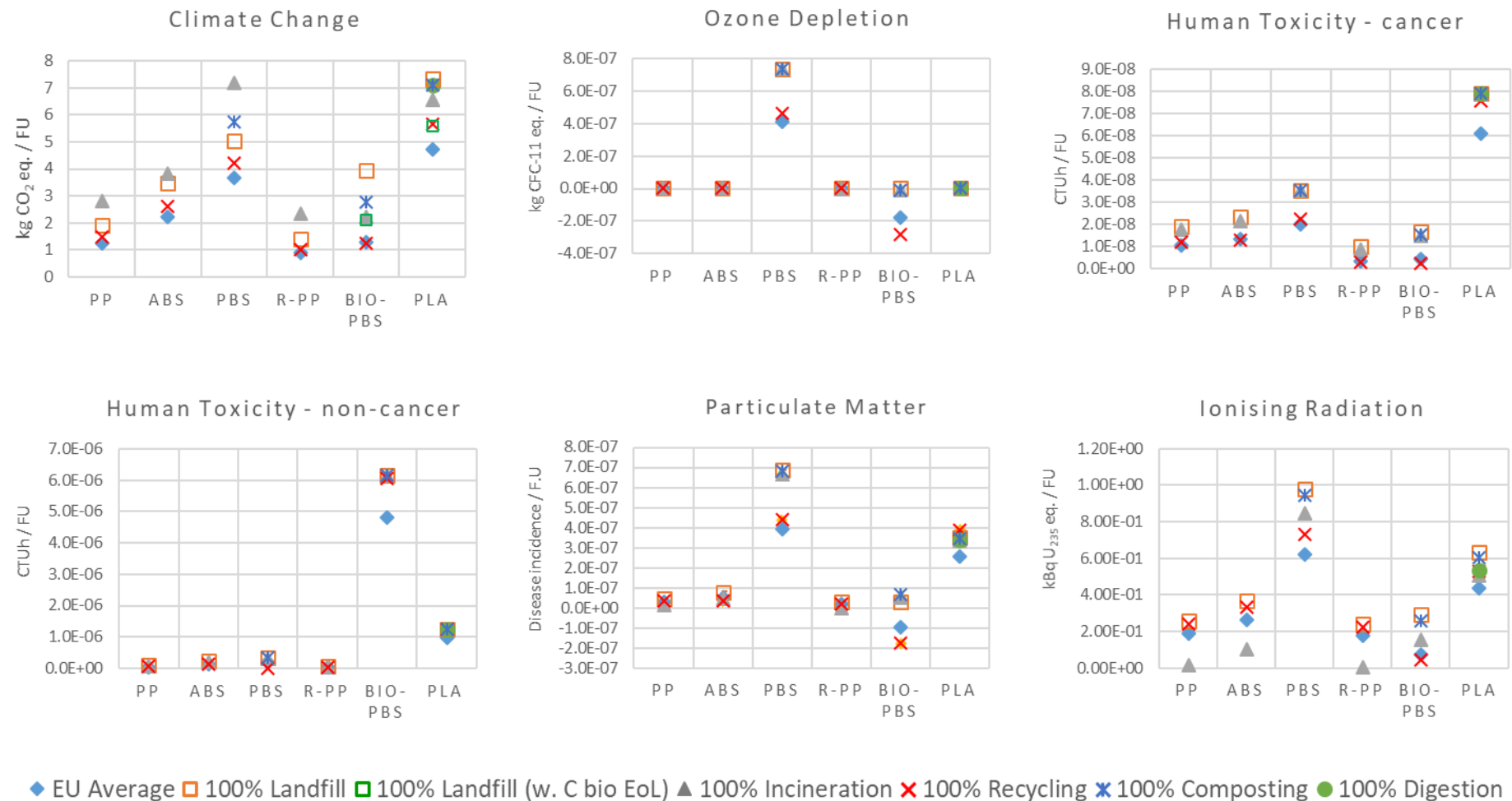
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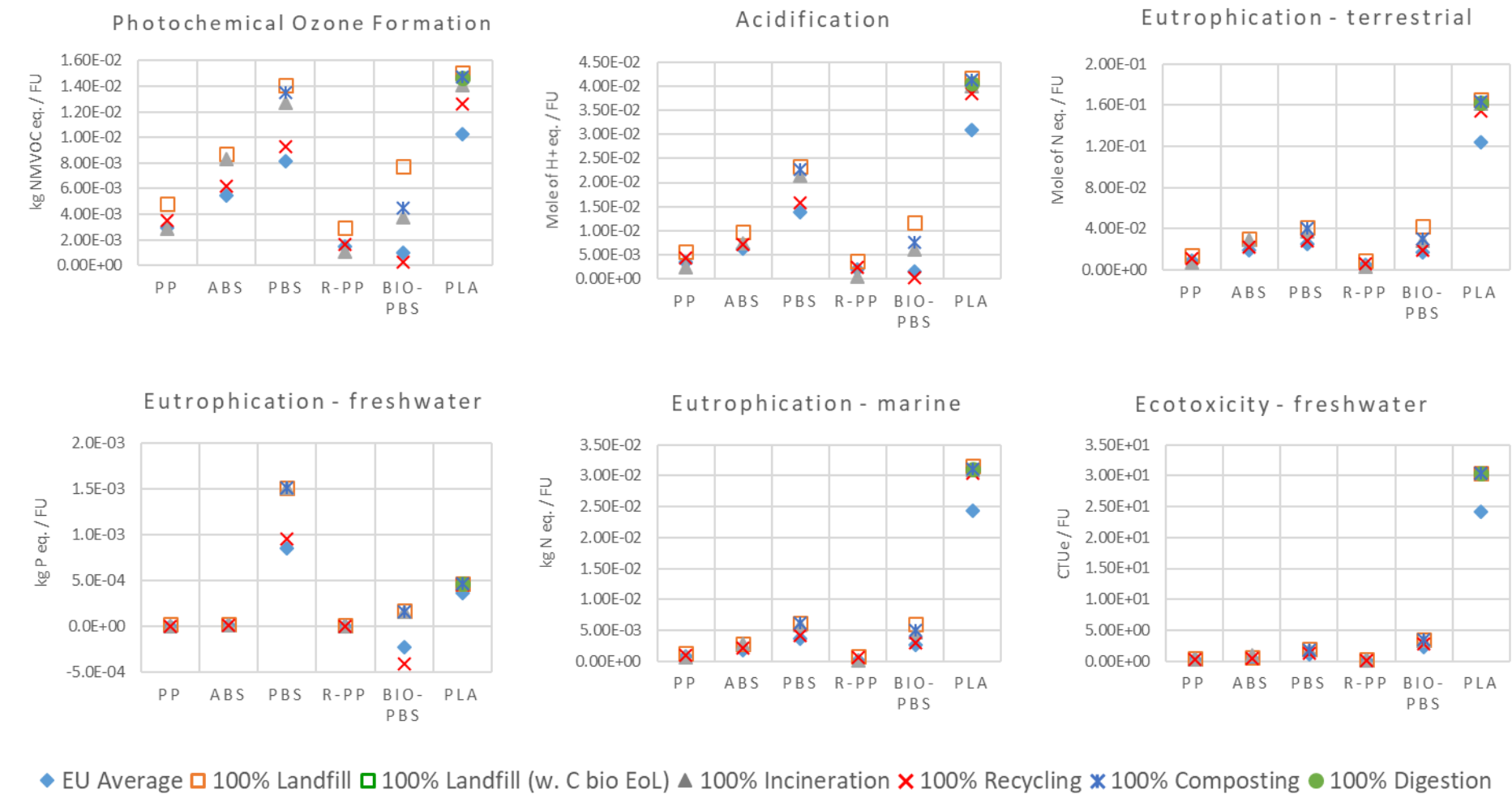
**Figure 10.9.** Potential impact of automotive interior panels LCA scenarios for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils. Note that in some impact categories a part of the results is out of scale and is curtailed.



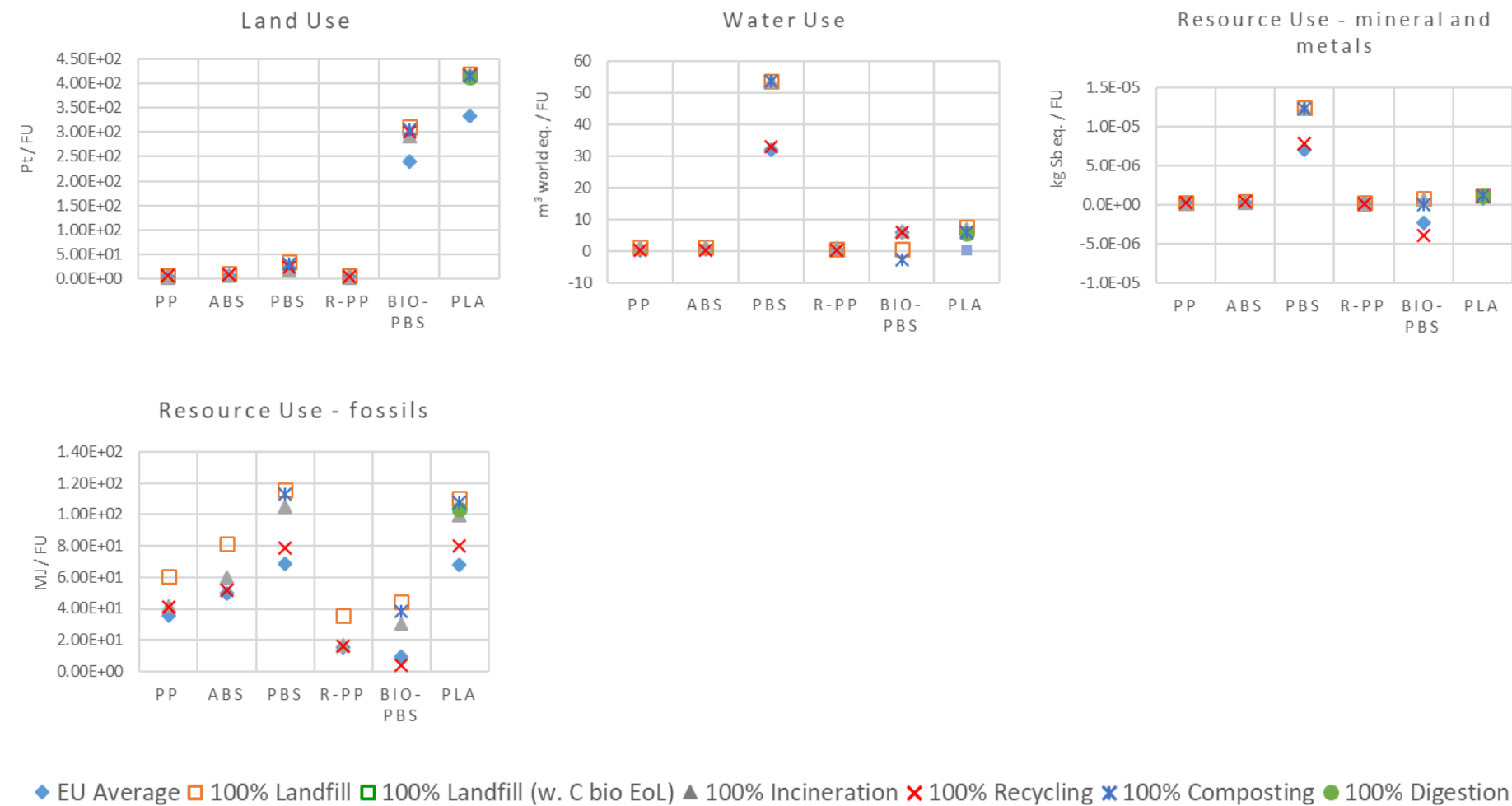
**Figure 10.10.** Contribution of fossil, biogenic, and land use/land transformation GHG emissions to the total Climate Change impact of automotive interior panels LCA scenarios (note that the contribution of iLUC to land transformation emissions is excluded).



**Figure 10.11.** Potential impact of automotive interior panels LCA scenarios for different End of Life options, for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity – non-cancer, Particulate Matter and Ionising Radiation. The Climate Change impact of the 100% landfilling scenario denoted with “C bio EoL” accounts for the contribution of biogenic carbon not released after 100 years from landfill deposition of bio-based automotive interior panels.



1 **Figure 10.12.** Potential impact of automotive interior panels LCA scenarios for different End of Life options, for the categories of Photochemical Ozone  
2 Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater.



**Figure 10.13.** Potential impact of automotive interior panels LCA scenarios for different End of Life options, for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils.



## 10.6 Additional Environmental Information

This section presents additional environmental impacts or aspects going beyond the default impact categories considered in the present method, which reflect official categories adopted in the PEF context. Additional environmental issues addressed include indirect Land Use Change (iLUC) impact on Climate Change, potential Biodiversity impacts, as well as the contribution of the investigated automotive panels scenarios to macro-plastics formation at End of Life (including product litter) and to micro-plastics generation throughout the supply chain.

### 10.6.1 iLUC impact

Table 10.11 presents the estimated contribution of GHG emissions from iLUC induced by bio-based feedstock supply to the potential Climate Change impact of bio-based automotive interior panel.

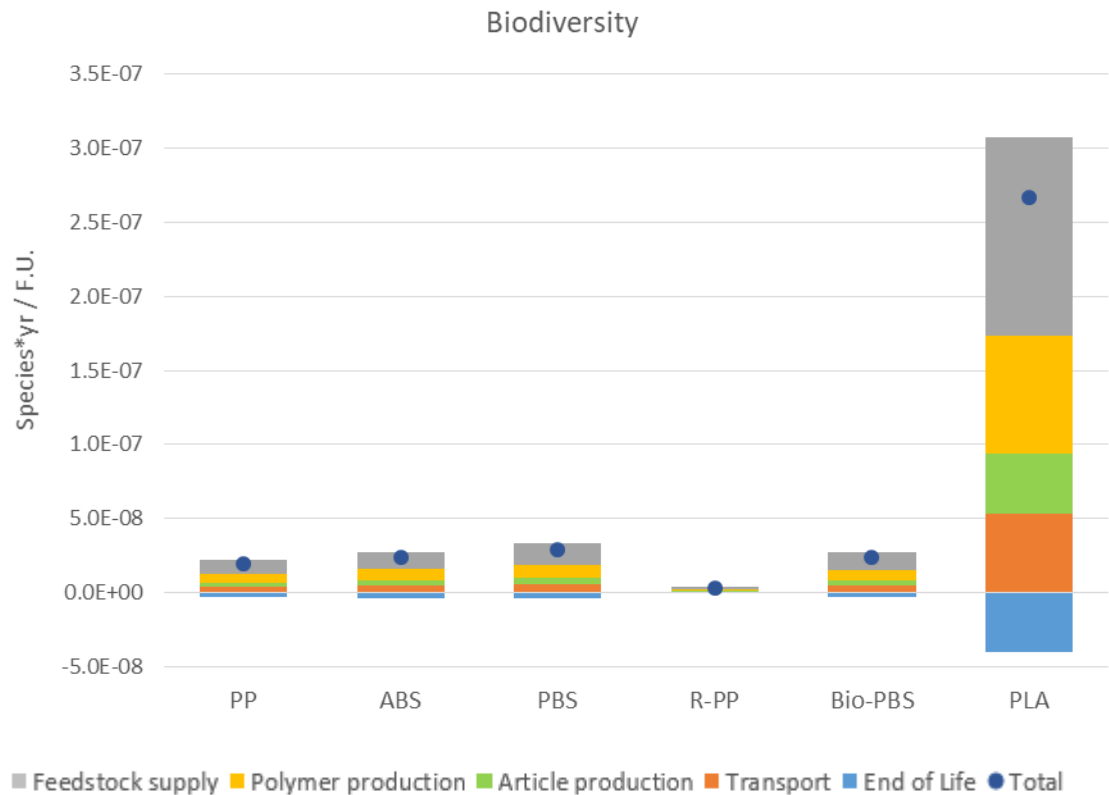
**Table 10.11.** iLUC contribution to the potential Climate Change impacts of automotive interior panel LCA scenarios.

Scenario / Polymer	iLUC Climate Change impact [kg CO <sub>2</sub> eq/FU]	Total Climate Change impact (incl. iLUC) <sup>(1)</sup> [kg CO <sub>2</sub> eq/FU]
S1 – PP	-	(1.27)
S2- ABS	-	(2.24)
S3 – PBS	-	(3.66)
S4 – R-PP	-	(0.88)
S5 – Bio-PBS	0.204	1.46 (1.28)
S6 – PLA	0.184	4.92 (4.72)

<sup>(1)</sup> Values in parenthesis refer to the Climate Change impact without the iLUC contribution.

### 10.6.2 Biodiversity impacts

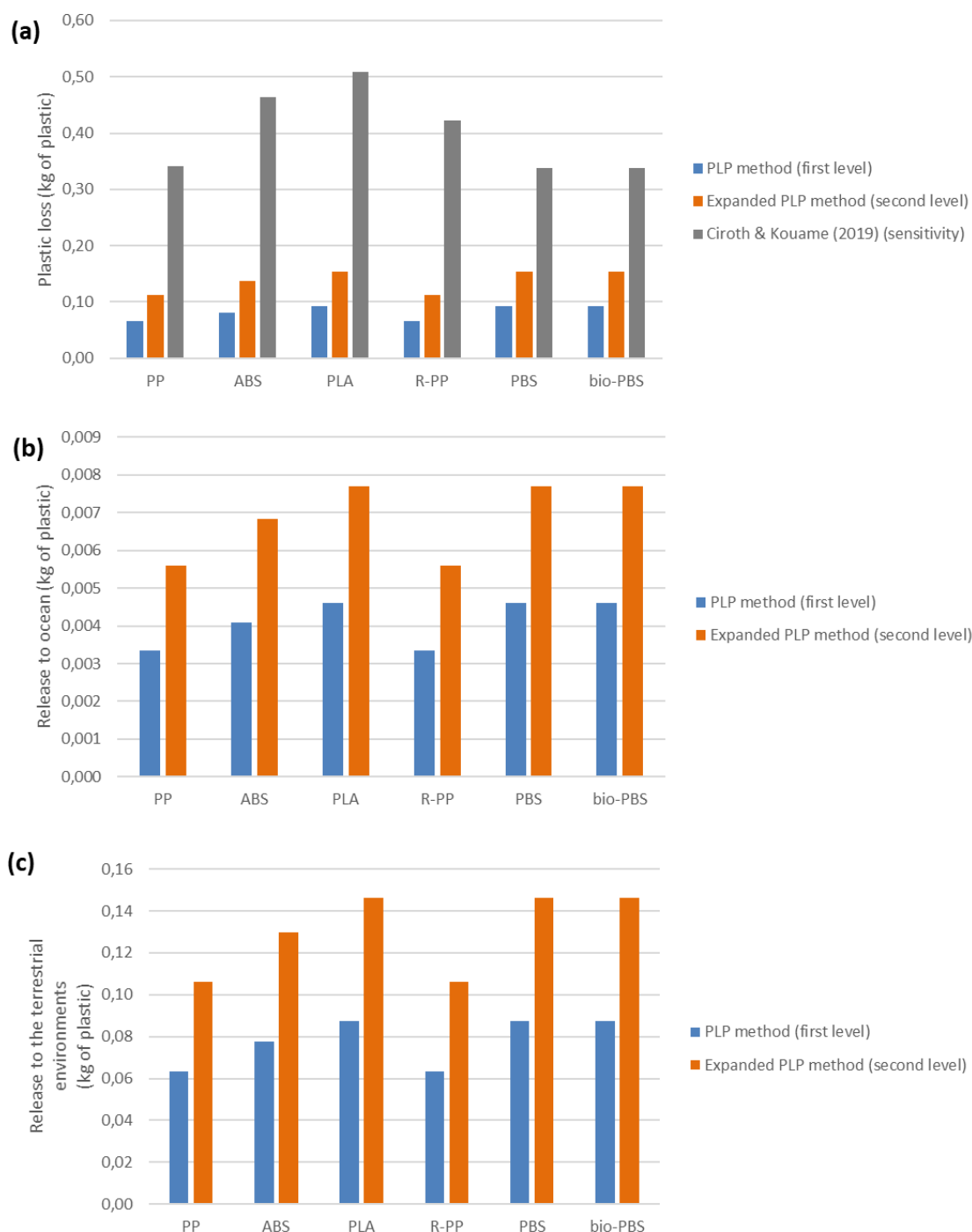
Estimated potential Biodiversity impacts of the investigated scenarios are presented in Figure 10.14. It is reminded that the assessment is performed through an endpoint-level indicator relying on impact assessment methods that for some of the underlying midpoint impact drivers differ from those recommended in the present method.



**Figure 10.14.** Potential biodiversity impact of automotive interior panels LCA scenarios, expressed as potential loss of animal species per year.

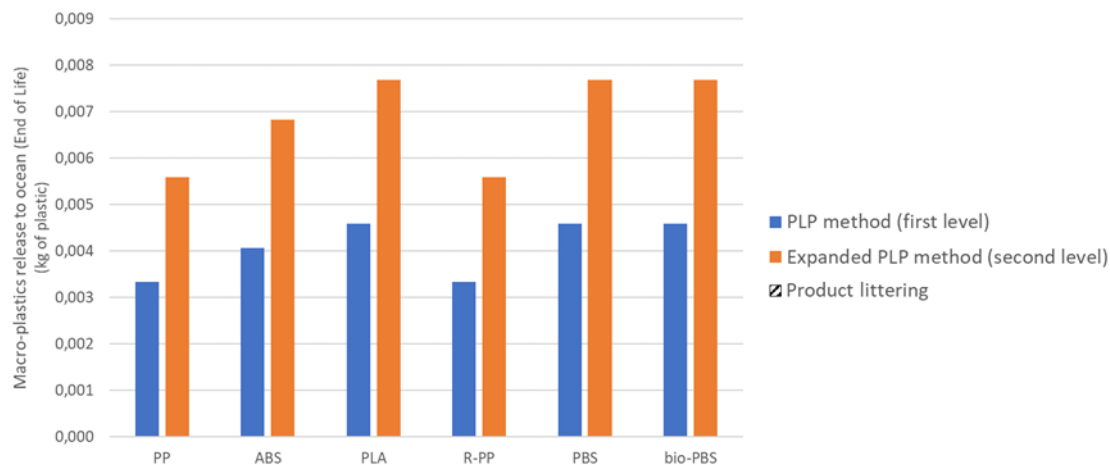
### 10.6.3 Macro- and micro-plastics generation

Figure 10.15 shows the total plastic loss, release to ocean and release to the terrestrial environment estimated for the assessed automotive interior panels scenarios, considering the whole product life cycle. Estimates obtained by applying all the three different explored approaches are presented, including the total loss and release to ocean/terrestrial environment calculated through both the *PLP method* (first level) and the *Expanded PLP method* (second level).



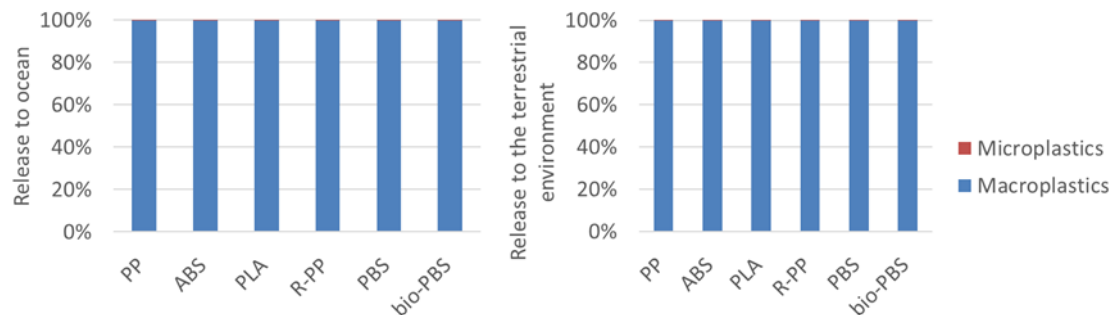
**Figure 10.15.** Total plastic loss (a), release to ocean (b) and release to the terrestrial environment (c) estimated for automotive interior panels LCA scenarios with different approaches. The contribution of both macro- and micro-plastics is included, considering the whole product life cycle.

Regarding the sole release of macro-plastics to ocean at End of Life (from product littering and waste mismanagement), Figure 10.16 presents the contribution of each automotive interior panels scenario, estimated with the two different explored approaches: the PLP method (first level), the Expanded PLP method (second level).



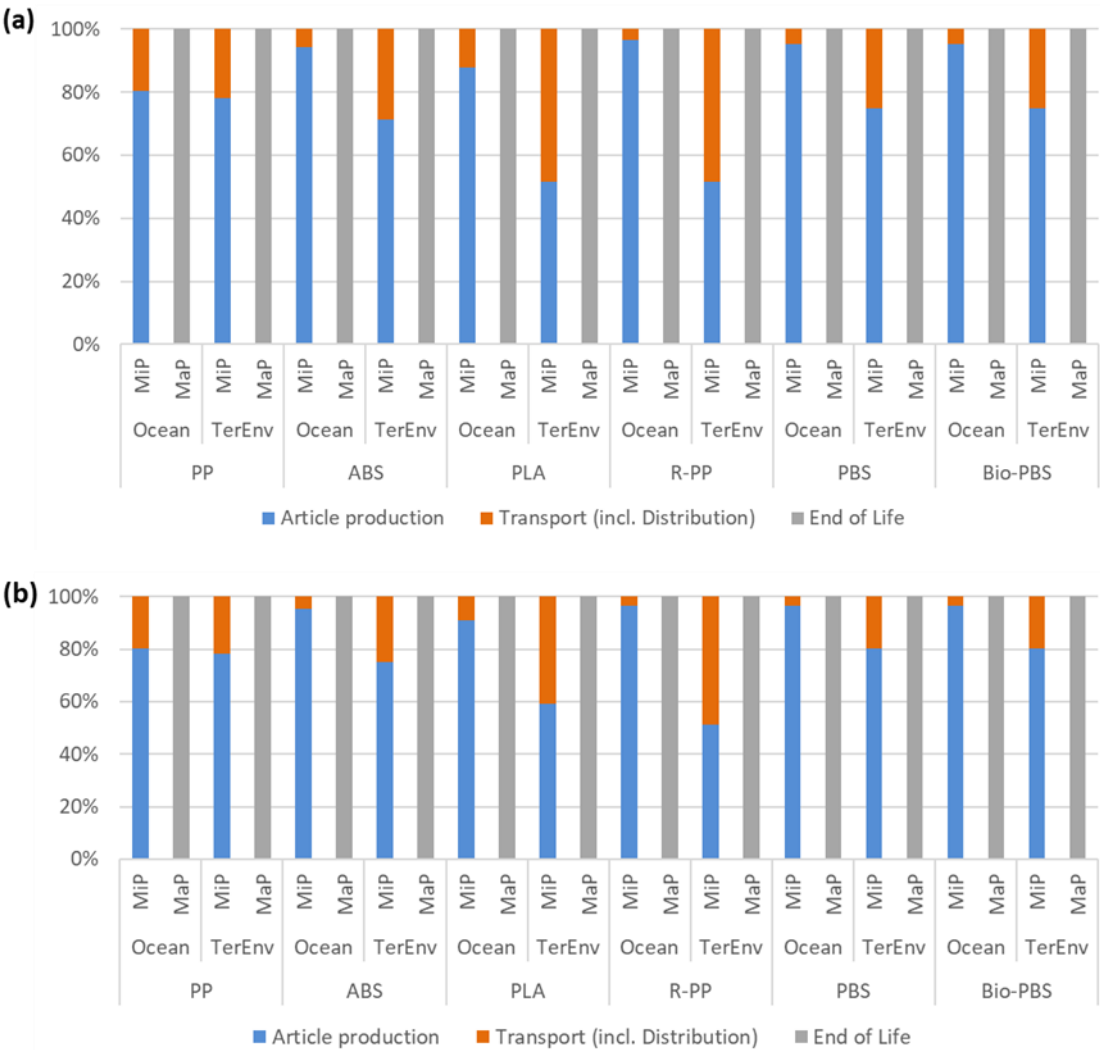
**Figure 10.16.** Macro-plastics release to ocean at End of Life, estimated with different approaches.

As for the share between the release of macro- and micro-plastics, Figure 10.17 shows as an example the relative contribution of the two categories to the total release to ocean and to the terrestrial environment estimated through the PLP method (first level). Note that the results are similar to those obtained when applying the Expanded PLP method (second level), as further discussed in Section 10.7.7.



**Figure 10.17.** Contribution of macro- and microplastics to the total plastic release to ocean and to the terrestrial environment estimated with the PLP method for each automotive interior panels LCA scenario.

The contribution of the different lifecycle stages to the total macro- and micro-plastics release to ocean and to the terrestrial environment of the assessed automotive interior panels scenarios has been finally evaluated. The results of this “hotspot analysis” are reported in Figure 10.18 for both the PLP method and the Expanded PLP method. Results are not shown for the other (sensitivity) approaches as they either cover only plastic losses or only a part of the product life cycle (i.e. End of Life).



**Figure 10.18.** Contribution of life cycle stages to the total macro-plastics (MaP) and micro-plastics (MiP) release to ocean (Ocean) and to the terrestrial environment (TerEnv) estimated for automotive interior panels LCA scenarios with (a) the PLP method (first level) and (b) the Expanded PLP method (second level).

## 10.7 Interpretation

In the interpretation of case study results, most relevant impact categories and lifecycle stages of the studied systems are firstly identified (10.7.1 and 10.7.2, respectively). Scenario impacts are then compared, considering both characterised LCIA results (10.7.3), and total normalised and weighted impacts (10.7.4). Results obtained by applying individually each viable End of Life options are separately discussed in Section 10.7.6, while the effects of iLUC and the contribution to macro- and micro-plastics generation are addressed in Sections 10.7.5 and 10.7.7, respectively.

Most relevant processes were not identified, since it was not possible to achieve a similar level of vertical disaggregation for foreground processes included in the life cycle models of the analysed scenarios. Therefore, the identification of most relevant processes cannot be carried out consistently across all scenarios, and a more detailed investigation for specific scenarios would not be meaningful. The identification of most relevant elementary flows was also not undertaken, as this would require prior identification of most relevant processes. Note, however, that any company, organisation or any other supply chain actor applying the present method shall proceed with the identification of both most relevant processes and elementary flows.

### **10.7.1 Identification of most relevant impact categories**

Table 10.12 shows the most relevant impact categories identified for each automotive panels scenario, based on normalised and weighted impacts, according to the approach outlined in the Method (Report I – Section 6.2.1). Relevant categories were identified as those that cumulatively contribute to at least 80% of the total normalised and weighted impact of the considered impact categories, excluding toxicity-related ones (these being still based on characterisation factors implemented in the EF 2.0 impact assessment methods, and hence not yet updated based on REACH data<sup>166</sup>). 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 as specified in Section 6.2.1 of the Method (Report I).

For fossil-based PP and ABS automotive panels, three impact categories are identified as the most relevant ones, i.e. Climate Change, Resource Use – fossils and Particulate Matter. These impact categories, are also within the most relevant ones for fossil-based PBS automotive panel, but the Water Use is the most relevant one. Climate Change, Resource Use – fossils and Particulate Matter are the most relevant categories also for recycled PP panels.

In the case of bio-based PBS automotive panels, Water Use, Climate Change and Land Use are the most relevant impact categories, while for PLA panels these are Climate Change, Resource Use – fossils, Water Use, Particulate Matter, Acidification and Eutrophication - terrestrial.

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<sup>166</sup> According to the latest version of the Product Environmental Footprint Category Rules Guidance (EC, 2018), toxicity-related impact indicators based on these characterisation factors shall be excluded from the procedure to identify most relevant impact categories.

**Table 10.12.** Most relevant impact categories identified for automotive interior panel LCA scenarios and related contribution to the total normalised and weighted scenario impact.

S1 – Fossil-based PP		S2 – Fossil-based ABS		S3 – Fossil-based PBS	
Impact category	Contrib.	Impact category	Contrib.	Impact category	Contrib.
Resource Use – fossils	45.7%	Resource Use – fossils	39.8%	Water Use	43.6%
Climate Change	34.1%	Climate Change	37.7%	Climate Change	18.2%
Acidification	4.0%	Particulate Matter	4.5%	Resource Use – fossils	16.4%
				Particulate Matter	10.2%
<b>Total</b>	<b>83.8%</b>	<b>Total</b>	<b>81.9%</b>	<b>Total</b>	<b>88.5%</b>
S4 – Recycled PP		S5 – Bio-based PBS		S6 – PLA	
Impact category	Contrib.	Impact category	Contrib.	Impact category	Contrib.
Climate Change	41.8%	Water Use	46.7%	Climate Change	29.8%
Resource Use – fossils	35.4%	Climate Change	35.7	Resource Use – fossils	20.6%
Particulate Matter	4.4%	Land Use	14.8%	Water Use	11.1%
				Particulate Matter	8.5%
				Acidification	8.2%
				Eutrophication – terrestrial	6.1%
<b>Total</b>	<b>81.6%</b>	<b>Total</b>	<b>97.2%</b>	<b>Total</b>	<b>80.9%</b>

### 10.7.2 Identification of most relevant life-cycle stages

Table 10.13 shows the most relevant lifecycle stages in the relevant impact categories identified in Section 10.7.1, and the associated contribution, quantified according to the rules detailed in the Method (Report I – Section 6.2.2). Most relevant stages include those that together contribute to at least 80% of the total characterised lifecycle impact in the specific category, and are highlighted in yellow in Table 10.13). For a more exhaustive picture, the contribution of all lifecycle stages is reported.

For fossil-based PP and ABS automotive panels, Feedstock Supply is the most relevant stage in Resource Use – fossils, followed by Polymer Production. In Climate Change the order is inverted: Polymer Production is the most relevant stage followed by Feedstock Supply. In Particulate Matter Polymer Production is followed by Article Production.

For fossil-based PBS automotive panels, Polymer Production dominates the most relevant life cycle stages in all relevant impact categories. This is also the case for bio-based PBS automotive panels. It is important to note that for the production of PBS polymer, data from the literature was used and therefore the feedstock supply could not be disaggregated. Also, because of the aggregated nature of the dataset for Bio-PBS manufacturing from US corn did not allow for Feedstock Supply disaggregation.

Climate change of recycled PP automotive panels is dominated by Polymer Production, while Resource Use – fossils and Particulate Matter are dominated by Feedstock Supply and Article Production, respectively.

1 In the case of PLA automotive panels, Polymer Production is the most relevant stage in  
2 Climate Change and Resource Use – fossils, followed by Feedstock Supply. For other  
3 relevant impact categories, Feedstock Supply is the most relevant one, followed by  
4 Polymer Production.

5 **Table 10.13.** Contribution of life cycle stages to the total characterised impacts of the most  
6 relevant categories identified for automotive interior panels LCA scenarios. Most relevant stages  
7 (i.e. those contributing to more than 80% of the total impact) are highlighted in yellow.

S1 – Fossil-based PP					
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	113.9%	Polymer Production	77.9%	Polymer Production	58.6%
Polymer Production	38.1%	Feedstock Supply	37.9%	Article Production	44.6%
Article Production	19.2%	Article Production	31.5%	Feedstock Supply	44.5%
Distribution	1.1%	Distribution	2.3%	Distribution	2.3%
End of Life	-72.4%	End of Life	-49.6%	End of Life	-50.0%
S2 – Fossil-based ABS					
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	88.1%	Polymer Production	110.7%	Polymer Production	97.9%
Polymer Production	66.9%	Feedstock Supply	23.2%	Article Production	30.3%
Article Production	16.8%	Article Production	21.7%	Feedstock Supply	26.1%
Distribution	1.0%	Distribution	1.6%	Distribution	1.6%
End of Life	-72.8%	End of Life	-57.3%	End of Life	-56.0%
S3 – Fossil-based PBS					
Water Use		Climate Change		Resource Use - fossils	
<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	177.5%	Polymer Production	147.4%	Polymer Production	155.7%
Article Production	0.2%	Article Production	15.0%	Article Production	13.6%
Distribution	0.0%	Distribution	1.1%	Distribution	0.8%
Feedstock Supply	0.0%	Feedstock Supply	0.0%	Feedstock Supply	0.0%
End of Life	-77.7%	End of Life	-63.5%	End of Life	-70.1%
Particulate Matter		Acidification			
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>		
Polymer Production	172.4%	Polymer Production	157.1%		
Article Production	4.4%	Article Production	12.2%		
Distribution	0.2%	Distribution	1.8%		
Feedstock Supply	0.0%	Feedstock Supply	0.0%		
End of Life	-77.0%	End of Life	-71.2%		



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S4 – Recycled PP					
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	86.6%	Feedstock Supply	129.5%	Article Production	70.0%
Article Production	45.3%	Polymer Production	55.5%	Polymer Production	57.7%
Feedstock Supply	27.3%	Article Production	43.7%	Feedstock Supply	35.1%
Distribution	0.9%	Distribution	2.6%	Distribution	1.3%
End of Life	-60.1%	End of Life	-131.3%	End of Life	-64.0%
S5 – Bio-based PBS					
Water Use		Climate change		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>
Polymer Production	115.2%	Polymer Production	192.1%	Polymer Production	127.4%
Article Production	1.2%	Article Production	42.9%	Article Production	1.8%
Distribution	0.0%	Distribution	0.8%	Distribution	0.2%
Feedstock Supply	0.0%	Feedstock Supply	0.0%	Feedstock Supply	0.0%
End of Life	-16.5%	End of Life	-135.8%	End of Life	-29.4%
S6 – PLA					
Climate Change		Resource Use - fossils		Water 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>
Polymer Production	111.5%	Polymer Production	128.9%	Feedstock Supply	107.8%
Feedstock Supply	30.8%	Feedstock Supply	18.9%	Polymer Production	11.2%
Article Production	11.6%	Article Production	13.7%	Article Production	1.2%
Distribution	0.9%	Distribution	0.8%	Distribution	0.0%
End of Life	-54.8%	End of Life	-62.3%	End of Life	-20.1%
Particulate Matter		Acidification		Eutrophication - terrestrial	
<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	83.0%	Feedstock Supply	86.5%	Feedstock Supply	95.3%
Polymer Production	45.9%	Polymer Production	41.2%	Polymer Production	33.4%
Article Production	6.7%	Article Production	5.4%	Article Production	2.9%
Distribution	0.3%	Distribution	0.8%	Distribution	1.1%
End of Life	-35.9%	End of Life	-34.0%	End of Life	-32.6%

2

### 3 10.7.3 Interpretation of characterised results

4 In this section, the characterised potential impacts of the assessed scenarios (Figures  
5 10.7-10.9 and Table B.7.7 in Annex B.7) are compared to evaluate the effects of using  
6 an alternative feedstock or material for a specific fossil-based polymer of reference used  
7 in automotive interior panel manufacturing. Therefore, the impacts of recycled PP

1 automotive panels are compared with virgin PP automotive panels. Since PP and ABS  
2 have different functions in the market and it is not clear whether PBS or PLA would  
3 replace one or another, the impacts of fossil-based PBS, bio-based PBS and PLA  
4 automotive panels are compared with both virgin PP and ABS automotive panels.

5 The comparison focuses on the sixteen, default impact categories considered in the  
6 Product Environmental Footprint (PEF) context and in this assessment. Potential impacts  
7 on Biodiversity are discussed separately, as being estimated via an endpoint indicator  
8 that is not recommended in the PEF framework, and partially relying on different impact  
9 assessment methods for the underlying midpoint-level impact categories. Note that, in  
10 the following comparison, differences between scenario impacts lower than 10% were not  
11 considered significant, in light of the uncertainty associated with the developed life cycle  
12 models and the applied impact assessment models. It has also to be noted that  
13 comparative considerations related to Human Toxicity and Ecotoxicity impact categories  
14 need to be interpreted in light of the higher uncertainty of the underlying impact  
15 assessment models and results. The same applies, to a lower extent, also to Water Use  
16 and Land Use.

17 Focusing on the relative performance of automotive panels made of the two fossil-based  
18 reference materials (i.e. PP and ABS), PP panel is found to be environmentally superior to  
19 ABS panel. PP automotive panel is preferable in all 16 impact categories, mostly due to  
20 the lower impact from polymer production and, to a much lower extent, article  
21 production and feedstock supply.

22 The alternative polymer 1, fossil-based PBS, has higher impacts in all categories  
23 compared to PP and ABS. For instance, bio-PBS has a Climate Change 2.9 and 1.6 higher  
24 than that of PP and ABS, respectively. Other impact categories are even higher, e.g.  
25 Ozone Depletion, with fossil-based PBS showing an impact that is 641 and 219 times  
26 higher than PP and ABS, respectively. This is due to the much higher impacts of the  
27 polymer production stage. It is important to note that the fossil-based PBS production  
28 was modelled using literature data; therefore these results should be carefully  
29 interpreted.

30 The use of recycled PP (alternative polymer 2) in panel manufacturing improves the  
31 environmental performance compared to virgin PP in all impact categories, which is  
32 mainly due to the much lower impacts in the polymer production and feedstock supply  
33 stages. For example, Climate Change is reduced by 31% compared to fossil PP, while  
34 Resource Use – fossils is reduced by 56%. It must be noted that the recycled content of  
35 the PP panel is 100% and that a lower recycled content would result in higher impacts  
36 compared to the 100% case.

37 Replacing fossil-based PBS with 100% bio-based PBS from US corn results in lower  
38 impacts for all categories but Human Toxicity – non-cancer, Land Use and Ecotoxicity -  
39 freshwater. The high impact of bio-based PBS in the Human Toxicity – non-cancer  
40 category (24 times higher than that of its fossil-based counterpart) may be due to the  
41 chemicals used in the polymer production stage. It is not clear if replacing PP with bio-  
42 based PBS is beneficial, since the impacts of bio-based PBS are higher in 7 out of 16  
43 categories. Among the categories in which bio-PBS has lower impacts two of the most  
44 relevant ones for PP, i.e. Resource Use – fossils and Human Toxicity – cancer. As for the  
45 third most relevant category for PP, Climate Change, this is virtually identical in both  
46 polymers. A similar trend is observed when bio-PBS is compared with ABS, although in  
47 this case Bio-PBS's Climate Change is 43% lower than in the ABS case.

48 Replacing virgin PP with PLA in automotive panels proves to be disadvantageous since  
49 this results in higher impacts for all categories. This is mainly due to the higher impacts  
50 in the polymer production and feedstock supply stages. A similar trend is observed when  
51 the results of PLA are compared with ABS, although in this case the bio-based alternative  
52 results in lower Ozone Depletion.

#### 10.7.4 Interpretation of normalised and weighted results

This section briefly compares the investigated scenarios based on the respective total impact scores, calculated by aggregating normalised and weighted impact assessment results across all impact categories (Table B.7.9 in Annex B.7). The comparison is carried out according to the same criteria outlined in Section 10.7.3 in terms of contrasted automotive interior panels scenarios, and considering significant only impact differences larger than 10%. While affected by greater uncertainty (from the calculation and application of normalisation and weighting factors), total impact scores provides a more immediate and synthetic indication of the overall environmental performance of the analysed product scenarios. This facilitates the interpretation compared to looking at characterised results from several impact categories. However, such estimates are inevitably affected by value choices underlying the definition of weighting factors aimed at establishing an order of relevance for the different impact categories in a European decision context.

The comparison among total normalised and weighted scenario impacts reflects the overall picture of results emerged when focusing on characterised impact assessment results in single impact categories, and can be summarised in the following points:

- Compared to fossil-based PP automotive interior panel, fossil-based ABS shows a decreased environmental performance with 60% higher total impact score.
- Recycled-PP panel shows an improved overall impact compared to its fossil-based counterpart, with a total impact score that is 43% lower than the latter.
- Biodegradable, fossil-based PBS shows a worsened overall impact compared to panel made of fossil-based PP with an impact score that is 5 times higher than the latter.
- Biodegradable, bio-based panels made of PLA shows a worsened overall impact compared to panels made of fossil-based PP with an impact score that is 4 times higher than the latter.
- Compared to panel made of fossil-based PP, biodegradable, bio-based panel made of Bio-PBS has an improved overall impact with a total score that is 4% lower than the former.

#### 10.7.5 Effects of indirect land use change (iLUC)

When the contribution of GHG emissions from iLUC is taken into account, only small or moderate changes are identified in the Climate Change impact of bio-based alternatives (i.e. Bio-PBS and PLA) (Figure 10.7 and Table 10.11). A maximum increase by 14% is observed for bio-based PBS panels, while for PLA panels the increase is equal to 4%.

Overall, accounting for iLUC effects does not affect the relative performance of the assessed scenarios (i.e. their comparison is not affected). However, this outcome needs to be read also in light of the GHG emission factors applied for iLUC in this assessment, which appear to fall in the lower end of the range of values available in the recent literature (see Section 4.4.16.3 in the Method (Report I)).

#### 10.7.6 Alternative End of Life options

This section discusses characterised scenario results calculated by individually applying each viable End of Life option as an exclusive (100%) End of Life scenario (as presented in Figures 10.11 to 10.13). The main purpose is to evaluate how the performances of the single automotive interior panel scenarios are affected by the change in the applied End of Life option, rather than to mutually compare such scenarios for different End of Life options. However, the results should not be interpreted as a strict comparison among different End of Life alternatives for the studied article, since the evaluation applies a product perspective, and burdens/benefits of selected End of Life options (e.g. recycling) are partitioned between different product life cycles (which prevents from capturing the

full implications of having a given waste stream routed to a specific End of Life option). In a waste management system perspective, where allocation is not needed, the total (system-wise) savings associated with the End of Life pathway “100% recycling” would likely be higher than what presented in this report where a product perspective is applied.

Because of the generally low contribution of the End of Life stage to the overall scenario impacts, none of the five considered product End of Life scenarios (i.e. mechanical recycling, incineration, landfilling, composting and anaerobic digestion) can be identified as preferable across all the assessed impact categories and automotive panel scenarios.

Comparing different EoL scenarios for a single material reveals that 100% incineration and 100% landfill are among the worse-performing scenarios, in line with the priority order outlined in the “Waste Hierarchy”, which sets them as the least preferable option (EC, 2008). 100% recycling is often the best option due the savings from the recycled material (i.e. avoided virgin material production).

The baseline scenario (current EU-average EoL for panels) is also performing well in many impact categories, including Climate Change, which can be explained by the large savings obtained at End of Life from reused panels.

### **10.7.7 Interpretation of results related to macro- and micro-plastics generation**

This section discusses the results presented in Section 10.6.3 on the estimated potential generation of macro- and microplastics of the assessed automotive interior panels scenarios. The latter are compared according to the same criteria applied to the interpretation of the potential environmental impacts (Section 10.7.3), initially focusing on the results from the two-level approach applied as a base case. The results from the approach applied as a sensitivity analysis to estimate total plastic loss are discussed separately.

Regarding the total loss of plastics (macro- and micro-plastics) and the resulting release to ocean and to the terrestrial environment (Figure 10.15), the two base-case approaches (*PLP method* and *Expanded PLP method*) provide similar results when comparing the different automotive interior panels scenarios. In both cases, the bio-based scenario of PLA is associated to a higher plastic loss and release to the environment compared to the respective fossil-based counterparts: PP (+37%) and ABS (+13%). This is mostly a consequence of the 37% and 13%, respectively, larger amount of polymer required per functional unit, which leads to a proportionally increased loss and release of plastics from both upstream life cycle stages (microplastics) and especially from product End of Life (macroplastics). Note, however, that the applied methods rely on mass-based indicators (which are affected by differences in panel mass) rather than quantifying the number of items (or plastic fragments/particles) lost or released to the environment. If this was the case, all scenarios would result in the same macroplastics generation at End of Life when applying both *PLP method* and *Expanded PLP method*, since panels with a lower mass would not imply a reduction in the contribution of product littering and waste mismanagement (which are based on release rates that are not affected by the type of feedstock or material). In turn, an identical macroplastics generation at End of Life would imply having a comparable total loss and release, these being mostly determined by macroplastics (see below).

Conversely, scenarios considering the use of recycled polymers show insignificant differences (<1%) in the total loss and release to fossil-based PP. As well, bio-based PBS show no different results than fossil-based PBS. This is because the pairs of scenarios require the same mass to fulfil the functional unit and the supply-chain includes similar transportation requirements. Such results highlight that changing the feedstock does not affect the dominating contribution from macroplastics loss and release at End of Life (see below for further discussion on this), while variations in microplastics generation due to

different transport requirements along the life cycle (reflecting different feedstock or polymer origins) only marginally affect the overall results.

Comparing the two base-case approaches, the *Expanded PLP* method results in a 67% higher plastic loss and release to ocean and the terrestrial environment compared to the original *PLP method* (Figure 10.15). This is mainly due to an increased contribution of macroplastics loss at End of Life, which also accounts for mismanagement of plastic waste exported for (sub-standard) recycling outside the EU (where the overall waste mismanagement share is increased by 5-6% compared to the assumption that all plastic waste is recycled in Europe). Conversely, the two additional microplastics sources considered in this alternative approach (road markings and marine coatings) does not significantly affect the results, due to the overall modest contribution of microplastics to the total release (in terms of mass).

The total supply-chain loss of plastics estimated based on the approach by Ciroth and Kouame (2019; sensitivity analysis) is between 3.6 and 6.3 times higher compared to applying the *PLP method*. This is likely a consequence of the broader range of processes covered (albeit in a more generic manner) compared to the *PLP method* (e.g. the contribution of background transport processes is also accounted for). Comparative results among the different automotive interior panels scenarios are similar to those obtained in the base-case assessment, with PLA panels showing the largest total loss<sup>167</sup>, and with no relevant differences when changing the type of feedstock used for PET or PBS panels production.

Focusing on the relation between total loss and release (which depends on the parameters of the *PLP method*), the plastic loss is modelled as fully released to the environment due to the lack of informal waste collection for this type of product. In relative terms, release to ocean accounts for a marginal share of the plastic loss (5%), while release to the terrestrial environment is the most relevant (95%). As shown in Figure 10.17 for the *PLP method*, the total plastic release to both ocean and the terrestrial environment is dominated by the release of macroplastics at the End of Life (more than 99%), while microplastics released throughout the upstream life cycle have only a minor role. Similar results are also observed for the *Expanded PLP method* (not shown), despite two additional sources of microplastics are considered in the latter (i.e. road markings and marine coatings). In both cases, this is a consequence of the prevailing mass of macroplastics, which is directly related to the mass of panels required per functional unit (reference flow), by means of higher loss and release rates compared to microplastics. Therefore, the mass of macroplastics is at least one order of magnitude higher than the mass of microplastics, which instead depends on the quantity of relevant lifecycle processes by means of (much) lower loss and release coefficients (depending on the source).

With regard to the origin of released plastics (in terms of life cycle stage and source), macro-plastics entirely originate from the End of Life stage (Figure 10.18), as they are only derived from mismanagement of the product as waste. Note that no littering is assumed for automotive interior panels as it is considered that panels are managed as part of cars at the End of Life thereby following specific collection and management pathways. On microplastics release, the relative contribution of the different sources (e.g. loss of plastic pellets and tire abrasion) depends on the characteristics of the supply-chain (e.g. feedstock origin, which affect the contribution from transport across the life cycle), as well as on the mass of polymer required per functional unit (which affect, to different extents, all the considered sources). However, within the limited role of microplastics in the overall release to the environment, the most relevant contribution is generally provided by pellet losses from the Polymer and Article Production stages, while transport-related sources have a more restricted role. Exceptionally, transport emissions are relevant (approximately 50%) for PLA and R-PP scenarios, whose supply-chains represent larger transport distances.

<sup>167</sup> The same comments as above on the implications of using a mass-based indicator need to be taken into account in the interpretation of this result.

1 In conclusion, due to the **dominant role of macro-plastics loss and release at the**  
2 **End of Life of automotive interior panels**, a best overall performance is associated to  
3 those scenario requiring a lower amount of plastic to fulfill the functional unit (i.e. relying  
4 on lighter automotive interior panels; mass-driven results). In this context, the use of  
5 **fossil-based and recycled PP automotive interior panels are the scenarios with**  
6 **the lowest plastic loss and release** to both ocean and the terrestrial environment. The  
7 same considerations apply to the sole release of macroplastics to ocean at End of Life,  
8 although in this case the use of an indicator based on the number of items (i.e. panels)  
9 released would provide the same result regardless of the scenario when applying the *PLP*  
10 *method*. Panels with different masses would indeed equally contribute to littering and  
11 waste mismanagement at End of Life. From a methodological perspective, the **different**  
12 **approaches provide similar results when comparing alternative scenarios.**

## 11 Case study 8: Printer housing panels

This case study focuses on injection-moulded panels used as exterior housing for large, multi-function office printers, which were assumed as a representative article of the use of plastics for enclosures of electronic (ICT) devices. The study was developed inspiring to, and further expanding on the preliminary work by Broeren et al. (2016), who conducted a screening life cycle assessment comparing the use of a number of (potential) alternative bio-based materials for printer panels manufacturing, with panels made of conventional polymers. However, the analysis was limited to GHG emissions, non-renewable energy use and land use as selected indicators for the comparison.

### 11.1 Assessed scenarios

The different polymers and the respective feedstock sources considered in the scenarios assessed for this case study are summarised in Table 11.1, which also specifies the End of Life options applied in each scenario.

Materials typically used for plastic enclosures of (larger) ICT equipment like office printers include ABS and blends of PC and ABS (Delgado et al., 2007; Broeren et al., 2016). PC/ABS blends are frequently (but not necessarily) used for more demanding applications, for instance when flame retardancy has to be ensured (such as in the frequently used PC-ABS-FR 40 polymer blend). PC typically has better mechanical and thermal properties than ABS, but higher costs, so that blending with ABS (which on the other hand improves the processing behaviour of PC) is performed, as far as this does not compromise the required mechanical and thermal performances for the intended application. High Impact Polystyrene (HI-PS) can also be used as a material for electronics housings, but normally for smaller appliances (e.g. data processing and consumer electronics, including household printers; Delgado et al., 2007). Therefore, virgin ABS and a PC/ABS blend (both derived from fossil-based feedstock sources) were considered as reference materials for the manufacturing of printer panels (Scenarios 1 and 2), which are also in line with the choices made in the screening assessment by Broeren et al. (2016). A 70/30 PC/ABS blend was specifically considered, according to the typical blend composition reported by Boyd et al. (2011) for laptop enclosures (i.e. 60-70% PC, 20-30% ABS and nearly 10% additives)<sup>168</sup>.

Recycled grades of both ABS and PC/ABS recovered from post-consumer WEEE<sup>169</sup> (and End of Life Vehicles in the case of ABS) are currently available on the market, with one major producer just being located in Europe (i.e. MGG Polymers GmbH). While their supply is not yet well established, and the recycling process still presents some technical challenges (especially for PC/ABS; Schwesig and Riise, 2016) it was considered relevant to also explore the use of both R-ABS and R-PC/ABS for printer panels production (Scenarios 3 and 4). Mixed plastics from the recovery of post-consumer WEEE (also known as Electronics Shredder Residue – ESR) were specifically considered as a feedstock, since ESR can be equally used to produce both types of recycled polymers, and also to reasonably limit data collection to the WEEE collection and recycling pathway (collection and pre-treatment routes of End of Life vehicles are different from those applied to WEEE). As a base case, and in the absence of more specific and representative data, a 35% recycled content was assumed in panels made of both materials, based on the ratio between the estimated amount of WEEE plastics currently sent to recycling in Europe and the total plastic demand for WEEE in the same region (i.e. 850,000 t vs 2.524.500 t)<sup>170, 171</sup>. However, since 100% recycled grades of both ABS and PC/ABS are

<sup>168</sup> Based on this data, an average composition including 65% PC, 25% ABS and 10% additives could be considered. However, since additives are not included as a base case in this assessment due to the lack of sufficiently robust and consistent data (as discussed further in this section), the share of additives was equally split between the two copolymers, leading to a 70/30 polymer blend.

<sup>169</sup> WEEE: Waste Electric and Electronic Equipment.

<sup>170</sup> An estimate of the recycling rate of WEEE plastics was hence used as an approximation for the recycled content (despite the two shares may differ in reality), in the absence of information on the actual uptake of (specific) recycled polymers in electronics applications.

starting to be available on the market (possibly with the use of proper additives to achieve suitable material properties; Schwesig and Riise, 2016), a 100% recycled content in panels was explored as a sensitivity analysis.

A possible bio-based alternative to fossil-based polymers (and especially ABS) is represented by the use of specific grades of (high heat) PLA resin, which are reported to deliver a performance comparable to that of PP and ABS when applied for consumer electronics enclosures (Total Corbion, 2019). This option was explored in Scenario 5, assuming the use of a crystalline PLA grade combining specific PLA types (i.e. P(L)LA with lower shares of P(D)LA). A similar alternative has been also considered in the screening assessment by Broeren et al. (2016). Maize cultivated in the US was assumed as a feedstock for PLA production, since nearly 90% of PLA currently consumed in Europe comes from the US (Eurostat, 2019a), where maize is used as a source of starch-derived sugar (dextrose) by the largest producer in the Country (Vink and Davies, 2015). The use of European maize was also investigated as a sensitivity analysis, along with PLA production taking place in the same region.

As a last alternative, the replacement of ABS with PLA as a blending copolymer in conventional PC/ABS polymer blends was considered (Scenario 6), based on the industrial experience reported in NaureWorks (2007)<sup>172</sup>, and on similar material alternatives explored in the already mentioned screening assessment by Broeren et al. (2016). A 65% PLA content was assumed for the polymer blend, which in the absence of representative data was estimated based on the biogenic carbon content reported for the different PLA/PC blends investigated by Broeren and colleagues (i.e. 32%) and the average carbon content in PLA (49.5%). Consistently with PLA used in the previous, fully bio-based scenario, US maize was considered as a feedstock for the production of this polymer, with both maize sourcing and PLA production in Europe being explored as a sensitivity analysis.

Regarding End of Life, as a base case the impacts of each scenario were assessed with reference to an EU-average End of Life scenario including all the options currently applied at the EU level to panels made of the specific material. In the case of panels made of materials not yet (widely) used for this type of applications (and for which no or only partly established treatment pathways exist at present, i.e. PLA and PLA/PC panels), those options that would be more likely applied in the current situation were considered in the EU-average scenario (taking also into account the need to ensure consistency with the scenarios applied to panels made of the reference materials they intend to replace). For all the investigated printer panels scenarios, the options included in the EU-average End of Life scenario were thus represented by mechanical recycling, incineration, and landfilling in Europe, as well as by sub-standard treatment and improper disposal outside the EU (for panels incorporated in -illegally- exported waste printers). These options were combined as described in Section 11.4.6.1, which also provides more details on each choice. In addition, scenario impacts were calculated by individually considering the application of each currently viable (or potentially viable) End of Life option, limiting to proper treatment/disposal options applied in the EU. Sub-standard or improper treatment and disposal outside the EU is, indeed, an option that should not be pursued, which should be progressively reduced, and which is thus not relevant to compare with the other proper End of Life scenarios. For panels made of PLA (which is a biodegradable material under controlled industrial conditions), viable options may also include biological treatments (i.e. composting and anaerobic digestion). These were not included in the EU-average End of Life scenario, since at present they are unlikely applied to PLA

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<sup>171</sup> The amount of WEEE plastics sent to recycling was estimated based on recent quantifications of WEEE flows collected for recycling in Europe for 2012 (4,250,000 t; Huisman et al., 2015), excluding the flow of steel-dominated appliances collected and processed as metal scrap, and assuming an average plastic content in collected WEEE equalling 20% (EERA, 2017). The demand of plastics for WEEE was based on the total polymer demand by converters for the Electrical and Electronic sector reported by PlasticsEurope for the same year as above, i.e. 2012 (PlasticsEurope, 2013).

<sup>172</sup> A 50/50 (flame-retarded) PLA/PC blend was reported to be developed by Fujitsu and Toray as a material for notebook housings (Fujitsu, 2005).



components in electronic devices (see Section 11.4.6.1 for further discussion). However, it was considered relevant to assess such treatment alternatives as individual (100%) options, to evaluate how the performances of the PLA printer panels scenario are affected compared to considering the other viable End of Life options.

**Table 11.1.** LCA scenarios assessed for the printer housing panels case study.

Scenario	Polymer	Monomer or Co-polymer	Feedstock	End of Life options <sup>(1)</sup>
1 - Conventional polymer 1	ABS	Acrylonitrile Polybutadiene Styrene	Fossil-based (crude oil/natural gas)	Recycling Incineration Landfilling Extra-EU treatment or disposal <sup>(2)</sup>
2 - Conventional polymer 2	PC/ABS blend <sup>(3)</sup>	PC ABS	Fossil-based (crude oil/natural gas)	Recycling Incineration Landfilling Extra-EU treatment or disposal <sup>(2)</sup>
3 - Alternative polymer 1	R-ABS (35% recycled content)	Acrylonitrile Polybutadiene Styrene	Waste ABS (from post-consumer WEEE)	Recycling Incineration Landfilling Extra-EU treatment or disposal <sup>(2)</sup>
4 - Alternative polymer 2	R-PC/ABS blend <sup>(3)</sup> (35% recycled content)	PC ABS	Waste PC/ABS (from post-consumer WEEE)	Recycling Incineration Landfilling Extra-EU treatment or disposal <sup>(2)</sup>
5 - Alternative polymer 3	PLA (crystalline, high heat)	Lactic Acid	Maize (US)	Recycling Incineration Landfilling Extra-EU treatment or disposal <sup>(2)</sup> Composting <sup>(4)</sup> Anaerobic digestion <sup>(4)</sup>
6 - Alternative polymer 4	PLA/PC blend <sup>(5)</sup>	PLA PC	Maize (US) Crude oil/natural gas	Recycling Incineration Landfilling Extra-EU treatment or disposal <sup>(2)</sup>

<sup>(1)</sup> The impacts of scenarios were individually assessed for each listed End of Life option (except for extra-EU treatment or disposal), as well as for a combination of such options reflecting as far as possible the current (real or potential) average situation at the EU level (except for composting and anaerobic digestion of PLA, which were only assessed as individual options).

<sup>(2)</sup> Included only in the EU-average End of Life scenario, but not assessed as individual (100%) End of Life option for the respective scenario.

<sup>(3)</sup> The blend includes 70% PC and 30% ABS (see the main text for explanations). The same share of the two copolymers was considered for both virgin and recycled PC/ABS blend.

<sup>(4)</sup> Assessed only as individual (100%) End of Life options for the PLA printer housing panels scenario.

<sup>(5)</sup> The blend includes 65% PLA and 35% PC (see the main text for explanations).

## 11.2 Functional Unit and reference flow

The main function of the studied article is to provide external housing for multi-function office printers. The functional unit was thus defined as “*providing 1 m<sup>2</sup> of external housing to a multi-function office printer for 10 years in the EU, without breaking, thermally deforming, and showing limited deflection during printer use*” (Table 11.2).

**Table 11.2.** Definition of the functional unit for printer housing panels LCA scenarios.

Aspect	Description
“ <b>What</b> ” (function(s) or service(s) provided)	Providing external housing for a multi-function office printer
“ <b>How much</b> ” (extent of the function(s) or service(s))	1 m <sup>2</sup> of external housing
“ <b>How well</b> ” (expected level of quality)	Without breaking and showing limited deflection under a minimum specified bending load, as well as without deforming after exposure to specified temperatures during printer use
“ <b>How long</b> ” (duration/lifetime of the function or service)	10 years
“ <b>Where</b> ” (location/geography of the service)	In the EU

The ability of the examined printer panel materials to achieve the mechanical (and thermal) performances required in the functional unit (i.e. withstanding a specified bending load and temperature without breaking or excessively deforming) mainly depends on the respective technical properties, which differ from one material to the other. Using different materials may thus imply having panels with a different mass (and hence a different reference flow) to ensure a comparable technical performance, as far as this can be achieved by increasing the panel thickness over its specific extension (as it is the case for the performance requirements considered in this study).

No immediate deterministic relationships are available to associate relevant technical properties of a material with the thickness of the panel (and hence with the mass of material) needed to achieve a required technical performance. Therefore, to account for differences in relevant technical properties of the compared materials in the calculation of the reference flow<sup>173</sup>, the concept of Material Substitution Factors (MSFs) was applied. MSFs express the ratio between the mass of a given material and that of a reference material, needed to perform identically a given technical function. In this case, ABS was considered as a reference material, along with the technical performances achievable, based on the respective technical properties, in a panel with a 3 mm thickness (in line with the reference thickness assumed in Broeren et al. (2016)).

As already discussed in Section 5.2, MSFs can be calculated on the basis of so-called Material Indices (MIs). The latter were introduced by Ashby (1999) to facilitate initial material selection in mechanical design, and aims to score the performance of a material with respect to a given (mechanical) function/performance (e.g. strength or stiffness), based on relevant material properties. These typically include the density of the material and technical properties relevant to the considered function (e.g. tensile strength or Young’s modulus). Materials with the highest Material Index can perform the intended function with the minimum mass<sup>174</sup>. Therefore the reciprocal of the ratio between Material Indices of the reference material and of an alternative one, can be considered to represent the Material Substitution Factor, according to the following Equation:

<sup>173</sup> The reference flow is the amount of material (polymer) required to fulfil the functional unit.

<sup>174</sup> Note, however, that material indices do not express the mass of material needed to perform a given (mechanical) function, but only the performance of the material with respect to that function.

$$MSF = m/m_{ref} = MI_{ref}/MI$$

where  $m$  and  $m_{ref}$  respectively represent the mass of the alternative and of the reference material needed to perform the intended function, while  $MI$  and  $MI_{ref}$  represent the Material Indices of the same materials, with respect to such function.

MIIs take into account one relevant technical property at each time (beyond material density). Therefore, different MIIs (and the corresponding MSFs) can be calculated, based on the different performance requirements considered in the functional unit (i.e. bending strength or bending stiffness in this case)<sup>175</sup>. A limiting performance requirement (or design constraint) can normally be identified in the one delivering the highest MSF, and hence the highest mass of material required per functional unit, which should be selected to ensure that all the specified requirements are met (as far as these do not counteract each other)<sup>176</sup>. However, for illustrative purposes in this assessment it was assumed that a single performance requirement could be prioritised at the time (regardless of whether the other requirement is met or not), to independently evaluate how the comparative performances of the assessed scenarios are affected by each specific material property. As a base case, the need for the panels of not breaking during use under a minimum specified bending load (i.e. of ensuring a minimum bending strength)<sup>177</sup> was considered as the priority performance requirement to be fulfilled. The effects of considering the need of not significantly deforming under a minimum specified bending load (i.e. of ensuring a minimum bending stiffness<sup>178</sup>) were instead explored in a sensitivity analysis.

The Material Index reported by Ashby (1999) when strength is the priority (or limiting) function for flat panels loaded in bending (with thickness as a free variable) is represented by the ratio:  $MI = \sigma_f^{1/2}/\rho$ , where  $\sigma_f$  is the flexural strength while  $\rho$  is the density of the material. This index was thus considered to calculate MSFs for the alternative printer panel materials investigated in this case study, and the corresponding reference flow. The main calculations carried out for this purpose, and the values assumed for relevant material properties are summarised in Table 11.3.

It is noted that the purpose here is to illustrate the concept of integrating relevant material properties in the calculation of the reference flow, for a more appropriate performance-based comparative assessment. In this perspective, the study does not aim at providing accurate estimates of the potential impacts of each printer panel material when a given technical performance needs to be ensured. Therefore, the calculation approach adopted in this study is considered suitable, despite it provides estimates of material requirements that may not be necessarily reflected in real products on the market.

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<sup>175</sup> Which express the ability of a material to withstand specified bending loads without breaking (strength) or without significantly deforming (stiffness). No Material Indices are available, instead, to express the performance of a material towards heat resistance (showing no deformation).

<sup>176</sup> For instance, if having a thicker panel does not imply a worsened performance towards any other performance requirement.

<sup>177</sup> Bending (or flexural) strength represents the ability of a material to withstand a specified bending load without breaking.

<sup>178</sup> Bending (or flexural) stiffness represents the ability of a material to withstand a specified bending load without deforming.

**Table 11.3.** Calculation of the reference flow for printer housing panels LCA scenarios when bending strength is the prioritised performance requirement to be fulfilled (the calculation is based on material substitution factors).

Polymer	Density - $\rho$ (g/cm <sup>3</sup> ) <sup>(1)</sup>	Flexural strength (N/mm <sup>2</sup> ) <sup>(1)</sup>	MI ( $\sigma_f^{1/2}/\rho$ )	MSF (m/m <sub>ref</sub> )	Reference flow (kg/FU)	Thickness (mm) <sup>(2)</sup>
ABS R-ABS	1.05	64	7.62	1	3.15 <sup>(3)</sup>	3.00
PC/ABS R-PC/ABS	1.17	90	8.11	0.940	2.96	2.53
PLA (high heat, crystalline)	1.25	76	6.97	1.09	3.44	2.75
PLA/PC blend	1.26	80	7.10	1.07	3.38	2.68

<sup>(1)</sup> Derived from Broeren et al. (2016). Values of flexural strength are not reported as a function of thickness.

<sup>(2)</sup> The thickness of ABS panels (reference material) was assumed (and used to calculate the reference flow, see note 3). For panels made of the other materials, the thickness was backward calculated for checking purposes, based on the estimated reference flow and the assumed panel extension (1 m<sup>2</sup>). The estimated thickness values fall within a reasonable range and are comparable with the thickness of ABS panels assumed as a reference.

<sup>(3)</sup> For ABS panels (reference material), the reference flow is calculated assuming a panel thickness of 3 mm, in line with the reference value considered in Broeren et al. (2016). This thickness could not be specifically related to the assumed flexural strength, and it is noted as a partial inconsistency.

### 11.3 System boundary

In all scenarios, the system boundary was set in order to cover the most relevant stages and processes of the full product life cycle (cradle-to-grave perspective), as described below and depicted in Figures 11.1 to 11.6:

- *Feedstock Supply*<sup>179</sup> – covering extraction, transport and possible refining of crude oil and natural gas (fossil-based polymers), collection, transport and pre-treatment of WEEE plastics (recycled polymers), crop cultivation (bio-based polymers), as well as transport of these feedstock sources to downstream conversion processes (e.g. naphtha cracking, polymer recycling, wet milling of maize);
- *Polymer Production*<sup>180</sup> – covering all the activities associated with the conversion of feedstock materials into the relevant monomer(s) and final polymer, including any transport among these activities and final transport of polymer granulate to downstream manufacturing processes;
- *Article Production*<sup>181</sup> – including printer housing panels manufacturing through injection moulding of polymer granulates<sup>182</sup>;
- *Distribution* – including transport of assembled panels from the manufacturing and assembly site to distribution centres, and from these to final consumers<sup>183</sup>;

<sup>179</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

<sup>180</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

<sup>181</sup> Corresponding to the default stage of “Manufacturing” specified in the Method and in the PEF framework.

<sup>182</sup> In this case study, printer housing panels were assumed to be manufactured within the same facility where the different printer components are assembled into a final printing device. Therefore, no specific transport was considered to take place for panels between these two activities.

<sup>183</sup> While panels assembled into printers are transported in reality, transport of printer components other than panels was excluded in this case study, due to the exclusive focus on the panel life cycle, and not on the life cycle of printers as a whole.

- *End of Life* – covering collection, transport, recycling, incineration, or disposal of printer panels after use, including any avoided processes from virgin material or energy substitution.

A different nomenclature was applied for some life cycle stages compared to the default nomenclature specified in the Method (and in the PEF framework) to make it more relevant for the investigated supply chains and the project scope. Moreover, the default “Raw Material Acquisition and Pre-processing” stage was further split into two separate sub-stages (i.e. Feedstock Supply and Polymer Production), to allow disaggregating the impacts of feedstock supply from downstream conversion processes, and hence to better appreciate any differences among the use of different feedstock sources (in line with the project goal).

The assembly of panels with other printer components, and the Use Stage of (panels in) printers were excluded, as it was the life cycle of such additional components. This is because this study focuses on the panels life cycle (and on the function provided by them) and not on the life cycle and function(s) of any printing device where panels are used. Moreover, the burdens of these activities and stages can be considered identical in all the investigated scenarios, since the use of different panel materials or feedstock sources normally do not affect the burdens of the Use Stage (e.g. the energy consumption of a printer during its lifetime), of panels assembly (unless large variations occur in the panel mass<sup>184</sup>), nor the type of additional printer components. The mentioned activities can thus be excluded from a comparative assessment, as the one performed in this study.

On the other hand, it has to be noted that the mentioned exclusions may weaken the assessment if they are not adequately justified (e.g. if relevant differences exist among compared alternatives, which are not taken into account). Therefore, the Use Stage, assembly activities (and the life cycle of additional printer components) shall always be taken in full consideration for possible inclusion in a comparative LCA study on printer housing panels (or plastic components in printers), paying particular attention to any differences among the compared alternatives. Moreover, such stages shall always be included in LCA studies of printing devices, in line with system boundary requirements in the Method and in the PEF framework.

As a base case, additives were not included in the assessment, due to the lack of complete and consistent data on the use of additives for the production of printer panels, of the examined polymers and of electronics plastic housings in general, as well as on their release and fate over the product life cycle. This is acknowledged as a limitation of this study, as additive production can account for a non-negligible portion of 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 (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 plastics routed to biological treatments or subject to in-situ degradation).

Due to the generally important role of additives in electronics enclosures (where flame retardants are also used, depending on the application), a sensitivity analysis was performed in this case study to gain insight into the potential environmental relevance associated with the use of these substances (Section 11.7.7.5). The evaluation was based on currently available data and information (which are incomplete or missing, especially for recycled polymers) and introducing an even considerable number of assumptions, so that the obtained results need to be interpreted with caution. However, the main purpose of the analysis was to better understand the contribution of additives released into the environment to the overall lifecycle impacts of printer panels, rather than to compare the performances of the different investigated alternatives. Therefore,

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<sup>184</sup> Which may affect the energy demand of the assembly process.

- 1 while some room is given also to a few comparative considerations, the results must be
- 2 mainly read in this perspective.

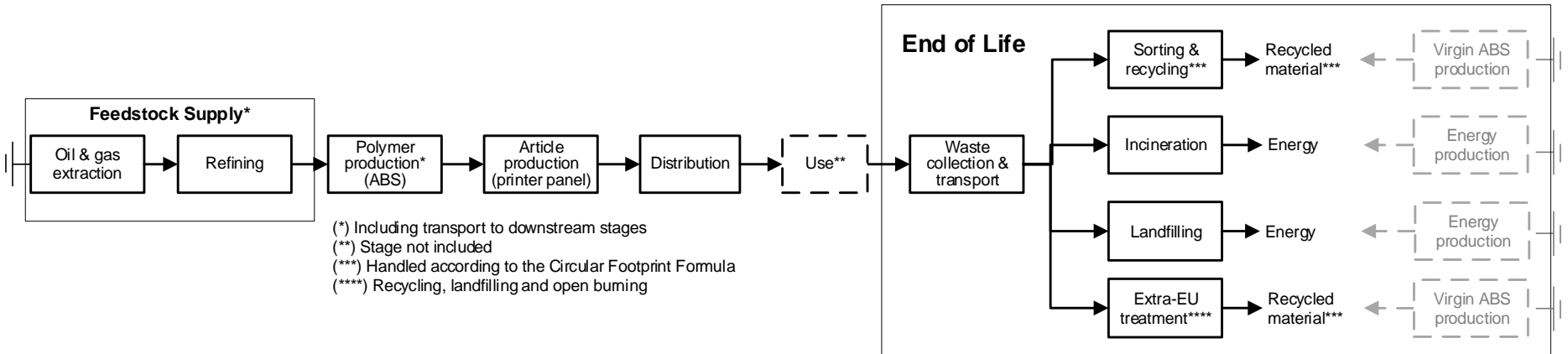


Figure 11.1. System boundary for virgin (fossil-based) ABS printer housing panels (Scenario 1).

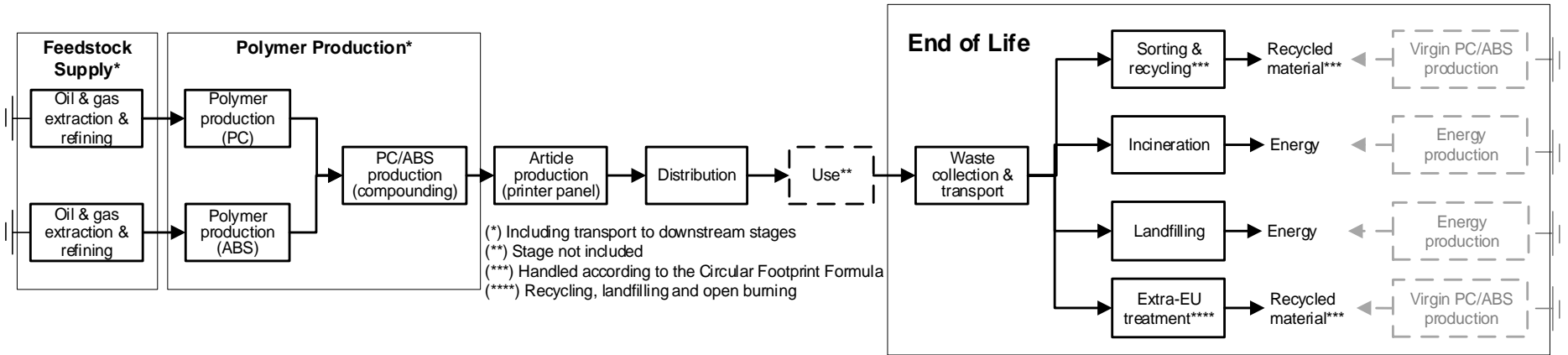


Figure 11.2. System boundary for virgin (fossil-based) PC/ABS printer housing panels (Scenario 2).

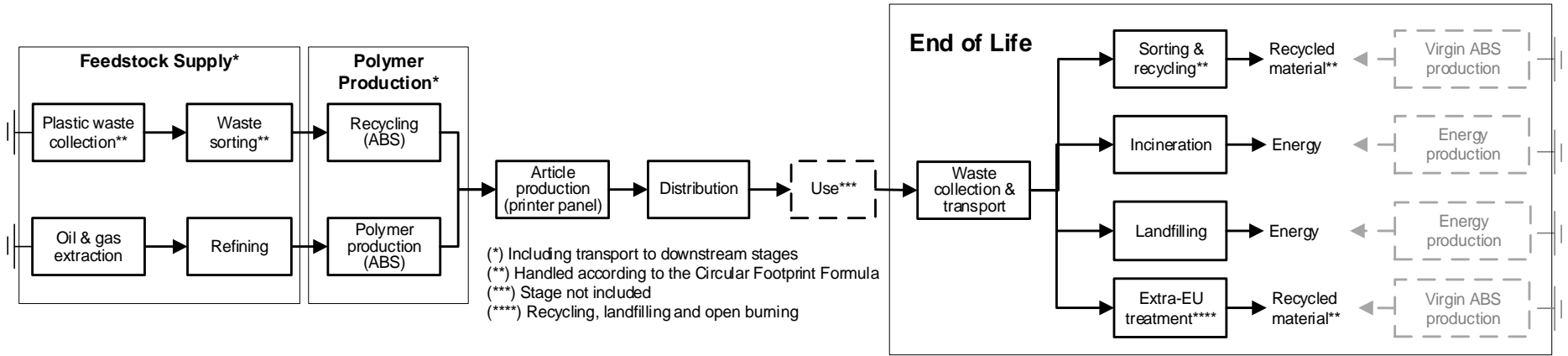


Figure 11.3. System boundary for 35% recycled ABS printer housing panels (Scenario 3).

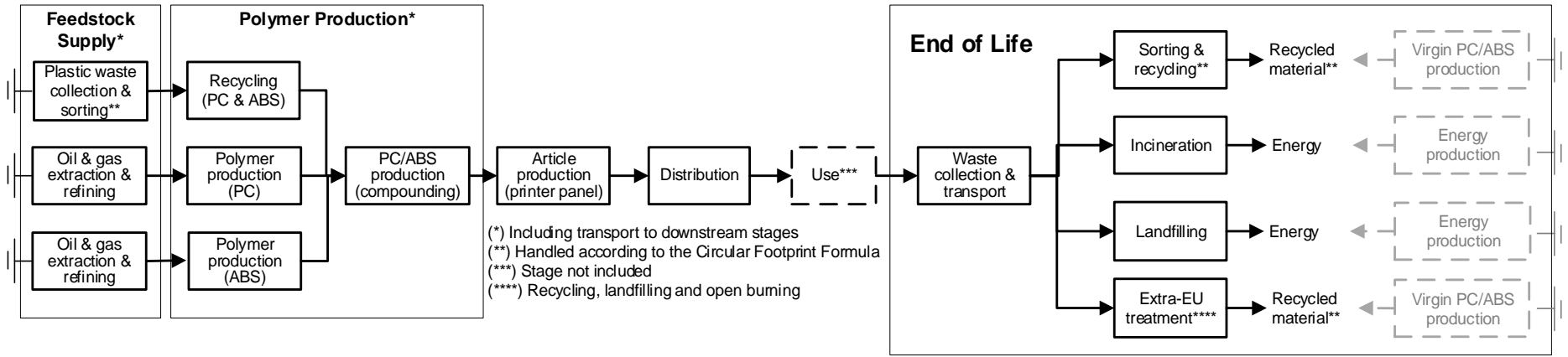


Figure 11.4. System boundary for 35% recycled PC/ABS printer housing panels (Scenario 4).



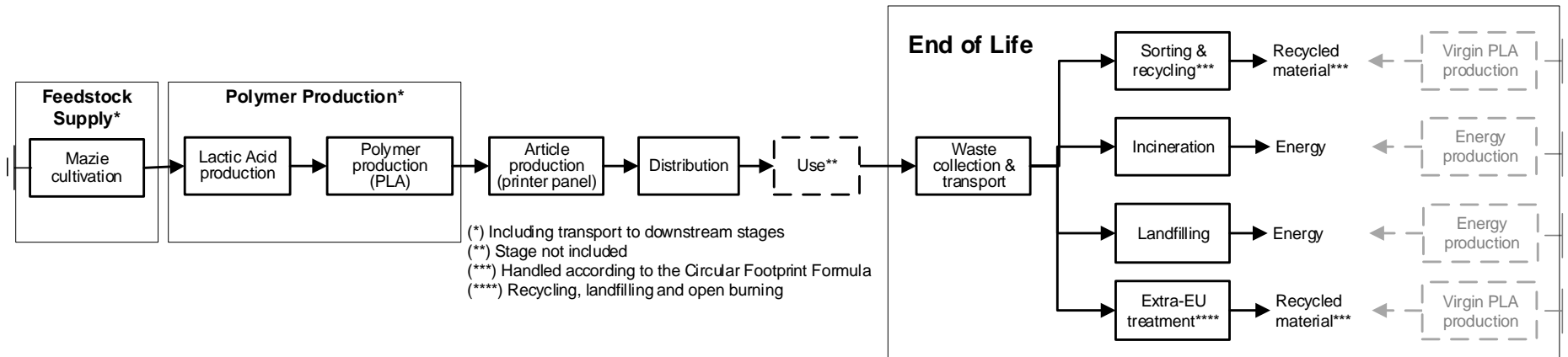


Figure 11.5. System boundary for PLA printer housing panels (Scenario 5).

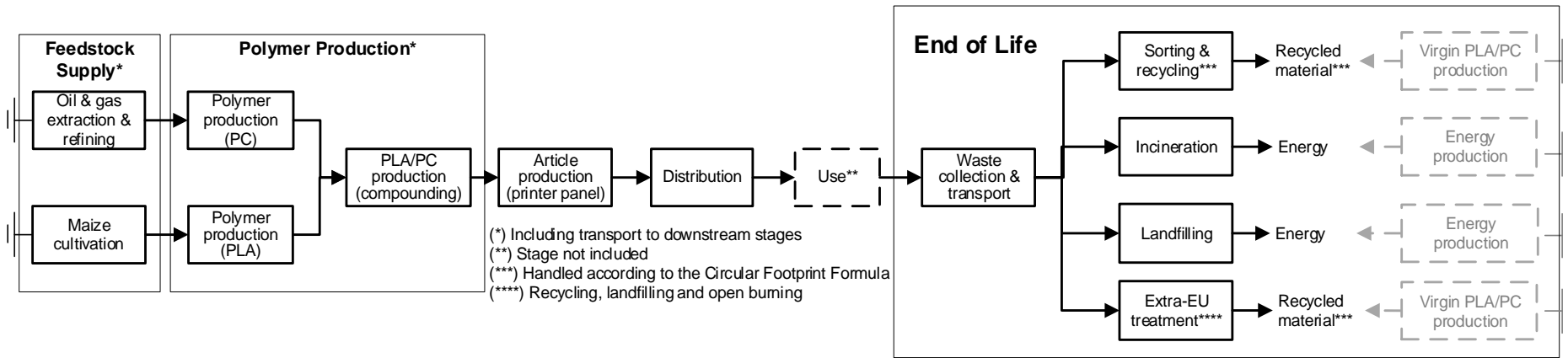


Figure 11.6. System boundary for PLA/PC printer housing panels (Scenario 6).

## 11.4 Life Cycle Inventory

This section describes the overall approach for developing the Life Cycle Inventory of the analysed scenarios, along with the related assumptions and data sources. The description is separated by major lifecycle stages, in the following sub-sections (11.4.1 – 11.4.7). The list of processes, related data sources, and main modelling details are provided in Tables B.8.1 to B.8.6 in Annex B8.

### 11.4.1 Feedstock Supply Stage

#### 11.4.1.1 Fossil-based polymers

For fossil-based polymers (ABS and PC), the stage of Feedstock Supply includes the processes of crude-oil and natural gas extraction and transport, naphtha production in crude oil refineries, and its subsequent transport to downstream conversion processes (typically naphtha cracking, or also catalytic reforming). The modelling of these activities was carried out as described in Section 4.4.1.1, where the reader is referred to for further details on data sources, covered sub-processes and related environmental burdens, main modelling choices (e.g. allocation) and any adjustments performed in the implementation of the applied datasets.

#### 11.4.1.2 Recycled polymers

For recycled polymers (R-ABS and R-PC/ABS), Feedstock Supply consists of collection of post-consumer WEEE plastic components made of the relevant polymer in Europe<sup>185</sup>, as well as its subsequent transport and pre-treatment in specific facilities to obtain a (bromine free) plastic-rich fraction for further separation and recycling. This plastic-rich fraction is typically referred to as Electronic Shredder Residue (ESR), and is obtained via dismantling and shredding (or direct shredding) of plastic-containing WEEE, followed by separation of any bromine-containing plastic fraction (through flotation). These activities and processes were modelled as described in Section 11.4.6 on End of Life modelling, and refer to separate collection and transport of plastic-containing WEEE at the regional level, as well as to their pre-treatment in dedicated industrial plants.

All the mentioned processes were implemented in the lifecycle model according to the Circular Footprint Formula (CFF), which is the default approach to handle recycling situations in the Product Environmental Footprint (PEF) context. According to this approach, only 50% of the burdens from collection and transport, pre-treatment, sorting and recycling were assigned to the recycled content in printer panels (assuming a situation of equilibrium between offer and demand of the two polymers, and hence setting the A factor equal to 0.5, similarly to the value recommended in the PEF context for other thermoplastic polymers like PE and PP). The remaining burdens were assigned to the system providing material for recycling (beyond the boundary of this case study). Further details on the implementation of the CFF are provided in Section 11.4.2.2 on recycled polymers production.

#### 11.4.1.3 Bio-based polymers

For bio-based polymers (i.e. PLA, used alone or as a blending copolymer with PC), the stage of Feedstock Supply includes cultivation of Maize in the United States, and its subsequent transport to further processing in the same country.

Maize cultivation was modelled through the EF-compliant dataset “[US] Maize (corn grain) production; technology mix | at farm”. This dataset is based on the approach used to develop agricultural inventories in the Agri-footprint database, relying on 5-year average yield data from FAOStat (2010-2014). Activities covered in the dataset are those of seeding and seed production, fertilizer and pesticide production and application, capital goods, as well as energy use and transport for field management practices. Harvested

<sup>185</sup> As part of the EEE (Electric and Electronic Equipment) waste stream.

maize was assumed to be transported to downstream processing along an overall distance of 100 km, covered by large lorries (> 32 t, fuelled with the US diesel mix).

#### **11.4.2 Polymer Production Stage**

The Polymer Production stage covers the activities of feedstock processing into relevant intermediates and monomer(s), the polymerisation process, possible compounding of polymer blends, as well as any transport among these activities and final transport of polymer granulate to the printer panels manufacturing site. The following subsections (11.4.2.1 – 11.4.2.4) describe how these activities have been modelled in the present case study.

##### **11.4.2.1 Fossil-based polymers and polymer blends**

For conventional, fossil-based polymers (ABS and PC), the whole process chain from feedstock processing to polymerisation, through the production of intermediates and monomers, was modelled by means of aggregated, gate-to-gate datasets provided by Thinkstep. Inputs includes combinations of crude oil, natural gas and naphtha, depending on the polymer. All conversion processes are assumed to take place in Europe, so that the datasets reflect EU average background conditions in terms of e.g. energy generation, material supply and transport.

For both polymers, one of the main conversion processes involved in the supply chain is steam cracking of naphtha and natural gas, delivering key intermediates for ABS co-monomers and PC precursors (Bisphenol-A and Phosgene). These intermediates include Ethylene (precursor in Styrene production), Propylene (used directly in Acrylonitrile production, and as a precursor in Styrene and Bisphenol-A production), Butadiene (directly used as a co-monomer for ABS), and other relevant intermediate flows (e.g. pyrolysis gas including mixtures of Benzene, Toluene and Xylenes). Other relevant conversion processes are catalytic reforming of naphtha and steam reforming of natural gas. In catalytic reforming, naphtha is processed, to produce Benzene, Toluene and Xylenes, with Benzene being used as an intermediate for Styrene and Bisphenol-A production. Steam reforming of natural gas generates synthesis gas consisting of Carbon Monoxide (directly used in Phosgene production for Polycarbonate synthesis) and Hydrogen, a precursor to Ammonia, which is used in Acrylonitrile production.

As for the steam cracking process of naphtha, the applied inventory is mainly based on industry data, completed, where necessary, by literature data. 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 co-products. 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 4.4.1.1), as the process typically takes place at refining facilities. In the subsequent separation process of reformat gas into its components (including Benzene used for Styrene and Bisphenol A 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. The final synthesis (polymerisation) process of ABS and PC is mainly modelled based on industry data from internationally adopted production processes, integrated with literature data when needed. No allocation is performed at this stage, being ABS and PC the only outputs of the respective production process.

In the case of polymer blends (i.e. PC/ABS), the compounding process (including blending of copolymers and granulation) was also modelled. In the absence of specific data, the burdens of this process were approximated with those of the compounding and granulation of recycled polymer flakes, as reported in the GaBi dataset "[EU-28] Pelletizing and compounding; technology mix | production mix, at plant | Pelletizing and compounding". The latter accounts for 2% material losses, which were assumed to be

recycled and handled through a cut-off approach<sup>186</sup>, to avoid unnecessary complication of the model by applying the Circular Footprint Formula to a minor material stream (which is not expected to significantly affect the results). The electricity input (which represents the only input of the process) was combined with the relevant EF dataset for average electricity supply from the grid in the EU.

Intermediate transport of copolymers from the polymerisation plant (inside or outside the EU) to the compounding site was also attributed to the Polymer Production stage, and modelled according to the default transport scenario specified in the PEF context (and in the present method) for transferring of goods from domestic or extra-EU suppliers to factories in Europe. This scenario is described more in detail in Section 11.4.2.4 (addressing transport of final polymer granulate to the article manufacturing site), and also accounts for transport of fossil-based polymers imported from extra-EU countries (i.e. ABS in this case).

#### **11.4.2.2 Recycled polymers**

Production of recycled ABS and PC/ABS blend implies recovering these polymers from the plastic-rich flow recovered from WEEE pre-treatment (Electronic Shredder Residue – ESR)<sup>187</sup>, through a process based on a number of sequential separation and purification steps. These may include further size reduction of ESR and removal of non-plastic materials (residual metals, wood, rubber and foam) via e.g. air classification and screening; separation of the resulting flow into plastic families with similar densities (consisting of one or more polymer types) and removal of most of the remaining contaminants via water-based density sorting systems (for both operations); further separation into homogeneous polymer streams (via electrostatic separation or froth flotation); as well as blending and final compounding/extrusion to granulates. The extrusion step includes possible compounding with additives, vacuum de-gassing, and melt filtration to remove any residual unmelt contaminant (Schwesig and Riise, 2016).

No inventory datasets representing the burdens associated with the recovery of specific polymers from the ESR stream are available, neither in the pool of EF-compliant datasets, nor in commercial databases. A recent LCA study has investigated the potential impacts of recovering a mixture of post-consumer plastics (including ABS, HIPS and PP) from ESR in a European plant owned by one of the major recyclers of WEEE plastics in the region (i.e. MBA Polymers; Wager and Hischier, 2015). However, the reported inventory is only partial (i.e. it only provides information on most relevant exchanges, but not on their quantity) and hence it could not be applied in this study for modelling purposes.

In the absence of specific and representative data, the burdens of both ABS and PC/ABS recycling from (bromine-free) ESR were thus approximated with those associated with the production of a generic, secondary plastic granulate out of sorted, post-consumer plastic waste. These were derived from 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", which was applied in the modelling. 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). 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).

<sup>186</sup> According to this approach, the burdens of the recycling process were entirely allocated to the system using the recycled material output (and hence not to the printer panels life cycle). At the same time, no benefits from recycling (e.g. avoided virgin material production) were allocated to the system supplying such recycled material (i.e. the printer panels life cycle).

<sup>187</sup> ESR is the plastic-rich residual material flow obtained from pre-processing of separately collected WEEE via dismantling, (manual) sorting, shredding and metal separation. ESR typically includes 85-90% plastic materials, combined with smaller shares of residual metals (e.g. wires), rubber, wood, foam, other non-plastic materials and fine particles lower than 3 mm (Schwesig and Riise, 2016).

It is acknowledged that the use of this dataset likely underestimates the impacts of the real ESR recovery process, which deals with a quite heterogeneous feedstock and, as described above, includes several separation and purification steps (applying a variety of technologies), while also performing melt filtration before extrusion to remove non-melts (Schwesing and Riise, 2016). Conversely, the modelled process deals with a homogeneous material input (albeit still metal-contaminated, in contrast to ESR), and relies on a simpler layout, including a reduced number of separation steps (i.e. metal separation and a "washing" process to remove small particles, dirt, and residues of oils and fats). Moreover, additives (e.g. modifiers and colorants) may also need to be used during the ESR recovery process, to adjust the (mechanical) properties of the recycled material to be as close as possible to those of the corresponding virgin polymer (especially when the use in more demanding applications is foreseen; Schwesing and Riise, 2016). However, additives were excluded from the base case assessment, due to the lack of reliable, representative and consistent data. Therefore, while additives are reported to be generally used only in small amounts (Schwesing and Riise, 2016), their exclusion contributes to further underestimate the potential impacts associated with the production of recycled ABS and PC/ABS. On the other hand, it must be noted that additives have been consistently excluded in all the investigated printer panels scenarios (including also those relying on virgin fossil-based polymers and bio-based polymers), so that their impacts are "symmetrically" affected by this omission (albeit not necessarily to the same extent). Finally, it must be also noted that the same recovery process was modelled for both ABS and PC/ABS, despite recovery of PC/ABS is reported to be more challenging, and hence potentially more energy-intensive. This is because (among others) the polymer is usually separated into an initial stream of plastics with similar densities, which is not easy to further separate into individual polymer streams. Therefore, additional separation and purification steps are normally required, potentially increasing the potential impacts of the process.

According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the recycling process were allocated to the recycled content in ABS and PC/ABS panels (assuming a situation of equilibrium between offer and demand of the two recycled polymers, and hence setting the A factor equal to 0.5, similarly to the value recommended in the PEF context for other thermoplastic polymers like PE and PP). However, the recycled content carries 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 life cycle providing the recycled material). Assuming a value of the  $Q_s/Q_p$  factor equal to 0.9 (similarly to the value recommended in the PEF context for other thermoplastic polymers, in the absence of more specific data for ABS and PC/ABS) the allocated share of virgin production impacts is equal to 45% ( $A \times Q_s/Q_p = 0.5 \times 0.9 = 0.45$ ). It is noted, however, that recycled ABS and PC/ABS are reported to achieve similar mechanical properties to the corresponding virgin polymers, possibly through the use of proper additives when the use in more demanding applications is foreseen (Schwesing and Riise, 2016). In this case, a substitution ratio ( $Q_s/Q_p$ ) equal or closer to 1 could be applied, provided that no other relevant material properties are negatively affected by recycling. However, due to incomplete knowledge about the relative quality of the two recycled and virgin polymers, and since additives possibly needed to achieve a suitable quality of the recycled polymer are unknown and could not be modelled, a lower quality was assumed for both recycled ABS and PC/ABS compared to their virgin counterpart (i.e. a lower share of virgin production burdens is taken up in the system). Virgin polymer production burdens were modelled as described above (Section 11.4.2.1) for conventional, fossil-based ABS and PC/ABS, and in Section 11.4.1.1 for the respective Feedstock Supply.

### **11.4.2.3 Bio-based polymers and polymer blends**

The only bio-based polymer considered in this case study is PLA, which is used either alone (in "neat" PLA printer panels), or as part of a polymer blend with Polycarbonate (in PLA/PC panels). For neat PLA printer panels, a particular PLA grade suitable for electronic

applications was specifically assumed to be applied, i.e. a crystalline PLA grade consisting mainly of P(L)LA combined with lower shares of P(D)LA.

Available life cycle inventories do not allow to differentiate between the production of “generic” (e.g. amorphous) and crystalline (or semi-crystalline) PLA. Therefore, the same aggregated, gate-to-gate dataset was applied in this study to model PLA production, regardless of the specific PLA type used for printer panels manufacturing. The dataset was provided by Thinkstep, and considers the use of maize as a feedstock, consistently with the PLA-based scenarios assessed in this case study. Processes covered in the dataset include maize wet milling for starch production, starch hydrolysis to glucose, its fermentation to lactic acid, oligomerization of lactic acid to lactide monomer, and final polymerisation of the latter to unspecified PLA. For the maize wet milling process, economic allocation is applied to the different co-products, while for downstream conversion processes no allocation is required. The overall inventory is mainly based on industry data from internationally adopted production processes, and is completed, where necessary, by literature data. It is noted that the dataset reflects European background conditions for energy and material inputs, despite the assessed printer panels scenarios assume PLA production taking place in the US and subsequent transport of polymer resin to Europe. This is acknowledged as a limitation of this study.

For the PLA/PC blend, the compounding process of the respective copolymers was also modelled as part of the Polymer Production stage, considering the same dataset, assumptions, and approach described for the PC/ABS blend in Section 11.4.2.1. Similarly, intermediate transport of the two copolymers from the polymerisation plant (inside or outside the EU) to the compounding site was also included in this stage, and modelled according to the default transport scenario specified in the PEF context (and in the present method) for transferring of goods from domestic or extra-EU suppliers to factories in Europe. This scenario is described more in detail in Section 11.4.2.4 (addressing transport of final polymer granulate to the article manufacturing site), and also accounts for transport of fossil-based and bio-based polymers imported from extra-EU countries (i.e. PLA in this case).

#### **11.4.2.4 Transport of polymer granulate to the article production site**

Modelling of transport of polymer granulate from the polymerisation or compounding plant (inside or outside the EU), to the printer panels manufacturing and assembly site in Europe, is based on the default transport scenario (distances and vehicle types) specified in the PEF context (and in this method) for the route supplier-to-factory. In the case of polymers produced in Europe (i.e. all the polymers and copolymers investigated in this case study except for PLA and the imported share of ABS), the following routes were thus considered:

- (i) 130 km by lorry (total weight >32 t; Euro 4);
- (ii) 240 km by train (average freight); and
- (iii) 270 km by ship (barge).

For the imported share of ABS (i.e. 25%), a transoceanic ship transport was considered as the main transport route. The corresponding overall 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)<sup>188</sup>. Countries contributing to at least 90% of the overall import were considered in the calculation, for an overall distance equal to 14,964 km (see Table 11.4). Oceanic ship transport was complemented with road transport to and from the harbour in both the exporting and importing country (i.e. the EU). Road transport is made by lorry (total weight >32 t; Euro 4) along an overall default distance of 1000 km.

<sup>188</sup> Available at: <https://www.searates.com/services/distances-time/>

A similar transport scenario to that of imported ABS was considered also for PLA. However, in this case, an overall sea distance equal to 6,000 km was assumed, based on the estimated mileage from the SeaRates calculation tool mentioned above for the suggested route New York – Rotterdam. A 1000 km road transport by lorry (>32 t; Euro 4) was again assumed for the route to and from the harbour in the exporting country (US) and in Europe.

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

**Table 11.4.** Calculation of the overall average sea distance for imports of ABS to Europe.

Exporting country	Import (¹) (%)	Import (% cum.)	Distance (²) (km)	Weighted distance (km)
KOREA, REPUBLIC OF (SOUTH KOREA)	70.1	70.1	16,702.31	11714
TAIWAN	16.8	87.0	15336.64	2583
THAILAND	4.9	91.9	13618.69	667
Other countries	8.13	100	-	-
Overall weighted distance				14,964

(¹) Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

(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 Middle-East and Asia (i.e. all countries contributing to ABS imports) were determined considering Marseille as destination port in Europe.

### 11.4.3 Article Production Stage

Regardless of the material and of the respective feedstock, printer housing panels were assumed to be manufactured through injection moulding, which is one of the most commonly applied process to convert raw polymer granules into rigid or semi-rigid plastic products (source). In the injection moulding process, melted or plasticised plastic material is injected by force into a clamped mould cavity (consisting of a single cavity or a number of similar or dissimilar cavities), where it is finally shaped into the desired object (Rosato et al., 2004).

The inventory of the mentioned process was derived from the aggregated, EF dataset "[EU-28+EFTA] Injection moulding; plastic injection moulding | production mix, at plant | for PP, HDPE and PE", which accounts for a 97.1% conversion efficiency. Process losses (e.g. flawed panels) were assumed to be entirely recycled in external facilities via re-granulation into new polymer pellets, ultimately replacing virgin granules of the same material. There are indeed examples of industrial operators collecting pre-consumer ABS and PC/ABS scraps from industry, and reprocess them into new polymer granules after (manual) cleaning, removal of any metal contamination and grinding into flakes. For consistency reasons, the same fate was assumed also for PLA and PLA/PC scraps, although it is unlikely to occur at present for such materials.

In the absence of specific data, the recycling process of manufacturing scraps was modelled based on an aggregated, EF-compliant dataset representing the production of a generic, secondary plastic granulate out of sorted, post-consumer plastic waste<sup>189</sup>. This process and the resulting virgin material substitution were implemented in the model

<sup>189</sup> "[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".

according to the Circular footprint Formula (CFF), as better described for End of Life recycling of post-consumer panels in Section 11.4.5.3 (where the reader is referred to for further details).

#### 11.4.4 Distribution Stage

Injection moulded printer housing panels were assumed to be manufactured within the same facility where the different printer components are assembled into a printing device ready to be delivered to the market. Therefore, no additional transport of panels between these two activities was modelled (and which, if modelled, should have been assigned to the Article Production stage).

On the other hand, transport of assembled panels from the manufacturing and assembly site to the final location of use of printing devices (i.e. private or public offices) was modelled as part of the Distribution stage. The modelling was based on the default transport scenario specified in the PEF context (and in this method) for the route *factory* → *distribution centres* → *final client*, considering the two following separate routes:

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

LCIs for all types of vehicles are available as EF-compliant datasets, which were used for modelling purposes.

#### 11.4.5 End of Life Stage

This section describes the modelling of the End of Life stage of printer housing panels. In particular, Section 11.4.5.1 focuses on the definition of the EU-average End of Life scenario, which is considered as a base case for the calculation of the potential impacts of the compared LCA scenarios. The remaining sections (11.4.5.2 – 11.4.5.8) address the modelling of waste collection and transport, and of the different End of Life options explored in the study. Finally, Section 11.4.5.9 provide case-specific details on the estimate of the potential contribution of printer panels to macro-plastics formation at the End of Life (including product litter) and to micro-plastics generation throughout the supply chain.

##### 11.4.5.1 End of Life scenario

At End of Life, printer housing panels follow the fate of the post-consumer electronic device they are part of, which in turn is a portion of the broader stream of Waste Electrical and Electronic Equipment (WEEE). The EU-average End of Life scenario for this article was thus defined based on the results of a recent reconstruction of WEEE flows in Europe (Huisman et al., 2015)<sup>190</sup>. Mass balance data and fate information available in this publication were combined as described in Table 11.5, leading to an overall End of Life scenario including the following pathways and corresponding rates:

- (i) 58.5% separate collection and recycling in Europe<sup>191</sup> (through sorting and reprocessing of the plastic-rich residual material fraction from WEEE pre-treatment and shredding<sup>192</sup>);
- (ii) 10.5% collection and treatment with residual municipal solid waste in Europe; and

<sup>190</sup> Note that no default, material-specific or application-specific values of the recycling rate (R2 factor) are available in Annex C to the PEF method. Therefore new estimates were generated based on available statistics, which were also applied to estimate the share of the other considered End of Life options.

<sup>191</sup> Including processing not necessarily compliant with the WEEE Directive.

<sup>192</sup> Referred to as Electronic Shredder Residue (ESR).



- (iii) 31% export and sub-standard recycling or improper disposal outside Europe  
(i.e. developing countries in Asia or Africa).

The rates estimated for these pathways are based on the simplifying assumption that all the WEEE flows reported in Huisman et al. (2015) contain the same share of plastic material (i.e. 20% on average, according to EERA, 2017), which is considered reasonable for the purpose of this study. In this perspective, the End of Life rates estimated for the WEEE stream as a whole could be identically applied also to plastics contained in WEEE (and hence also to printer panels) leading to the overall scenario reported above.

The fate of printer panels (and more in general of WEEE plastics) collected with residual municipal waste (10.5%) was defined according to official statistics on municipal waste management in Europe (Eurostat, 2019d), and included incineration (5.5%) and landfilling (5%). The relative shares of these two options were calculated as the average of the shares estimated based on Eurostat data for the years 2015-2017 (i.e. the three most recent years for which data were available), equalling 53% for incineration and 47% for landfilling.

As for the fate of (plastics in) WEEE exported outside Europe, no specific and representative data were available. Therefore, 50% of exported WEEE (i.e. 15.5% of total collected WEEE) was assumed to be processed and recycled in sub-standard conditions, the rest being improperly disposed via open burning or unsanitary landfilling. These options were applied in equal shares (i.e. to 25% of exported WEEE each, corresponding to 7.75% of total WEEE collected). For modelling purposes, export was assumed to take place to India, considering the availability of inventory data limited to sub-standard treatment (recycling) in such country (as better detailed in Section 11.4.6.6).

The same End of Life scenario described above was applied to all printer panels alternatives, including those based on emerging materials for this type of application (i.e. PLA and PLA/PC panels). Indeed, their fate largely depends on the fate of the WEEE stream, which includes pathways that in most parts are not affected by the type of material used for single WEEE components such as plastic enclosures. On the other hand, at present such emerging materials are typically not sorted out for recycling from shredded electronics residues from WEEE pre-treatment, where they are present in quantities not justifying this effort. However, it was here assumed that once their use would become more established, also their separation for recycling would become more advantageous and actually implemented. This assumption also prevents distorted comparisons with the other investigated printer panels scenarios, where a non-negligible recycling rate is assumed.

As briefly discussed in Section 11.1, while biological treatment options (including composting and anaerobic digestion) may also be viable alternatives for PLA panels, they were not included in the EU-average End of Life scenarios, since at present they are very unlikely applied to PLA parts in electronic devices. Indeed, these parts would need to be purposefully identified and (manually) separated from WEEE before shredding, and this is not considered feasible nowadays (at least for more advanced industrial facilities for WEEE pre-treatment/recovery). As an alternative, PLA flakes may be sorted out from shredded electronic residues after WEEE pre-processing, and then routed to biological treatments. However, after such non-negligible effort they would be most probably recycled, to keep the value of the material in the loop, rather than being composted or digested. Therefore, composting and digestion were only assessed as individual (100%) options for PLA printer panels, with the main purpose of evaluating how the performances of this scenario are affected compared to considering other viable End of Life options (including recycling, incineration and landfilling).

**Table 11.5.** Definition of the EU-average End of Life (EoL) scenario for printer housing panels, based on WEEE mass flows estimated for Europe in Huisman et al. (2015) and the related fate.

Fate of WEEE	Amount (kt)	%	Applied End of Life option (EU-average EoL scenario)	%
Official collection and recycling in EU	3300	45.5	Separate collection and recycling in EU (via sorting and reprocessing of plastic-rich Electronics Shredder Residues – ESR)	58.5
Non-compliant collection and recycling in EU <sup>(1, 2)</sup>	2200	-		
Additional non-compliant processing in EU <sup>(3)</sup>	950	13.0		
Collection with residual (municipal) waste in EU	750	10.5	Municipal incineration in EU	5.5
			Sanitary landfilling in EU	5
Exported (outside EU)	1500	20.7	Export and sub-standard treatment or improper disposal in Asia (India), including:	31
Scavenged and stolen parts <sup>(4)</sup>	750	10.3	<i>Processing and sub-standard recycling in the informal sector</i>	15.5
			<i>Open burning</i>	7.75
			<i>Unsanitary landfilling</i>	7.75

<sup>(1)</sup> Flow consisting of steel-dominated consumer appliances collected and processed under non-compliant and sub-standard conditions with other metal scraps.

<sup>(2)</sup> Not considered for the definition of the EU-average End of Life scenario of printer panels (which are not part of a steel-dominated appliance).

<sup>(3)</sup> Flow consisting of large IT equipment, professional heating and cooling installations, medical equipment, lamps (processed by installation companies; lamps collected in e.g. glass containers).

<sup>(4)</sup> Refrigerators, compressors, cables and IT components (commonly exported to Asia, predominantly as material fractions for further separation).

#### 11.4.5.2 Modelling of WEEE collection and transport

Collection and transport for recycling of waste panels in separately-collected post-consumer printers was modelled according to the pathways, vehicle types and distances identified in the study by Biganzoli et al. (2015) for the R4 WEEE category (including IT equipment, small household appliances, etc.). The data refer to a region with a well-developed waste management system in northern Italy, and can be considered representative of several regions in Europe relying on similar advanced waste management schemes. While this may not be the case for all European regions, the approximation is considered reasonable, due to the generally moderate contribution of waste collection and transport to the overall End of Life (and lifecycle) impacts (e.g. Biganzoli et al., 2015). Table 11.6 summarises the main details of the considered collection and transport pathways, which were modelled by means of EF-compliant datasets related to the use of vehicles for collection and transport.

Relevant data and assumptions for the modelling of collection of post-consumer panels in printers discarded as residual waste, and of their subsequent transport to incineration and landfilling, were derived from Rigamonti et al. (2013). These refer, again, to a region with a well-advanced waste management scheme in northern Italy, and were implemented in the model as detailed in Table 11.7. In this respect, it has to be noted that when EF incineration and landfilling datasets were used, no transport of collected waste was separately modelled, since the respective burdens are already accounted for in such datasets. In this case, only collection was modelled, according to the approach described above.

For panels collected with post-consumer printers exported to extra-EU countries (in this case India, as described in Section 11.4.5.1), the same collection pathways considered for panels in printers separately-collected for recycling in the EU were assumed and modelled (Table 11.6). Conversely, the subsequent transport outside the EU was

1 modelled according to the default transport scenario specified in the PEF context (and in  
2 the present method) for imports of goods from extra-EU suppliers (although applied in  
3 this case to an opposite situation of export). This scenario includes road transport from  
4 collection centres/platforms to harbours in Europe along a default distance of 1000 km  
5 (covered by lorries >32 t, Euro 4); transoceanic ship transport to India along a specific  
6 distance of 10.267 km, and final road transport by lorry (>32 t, Euro 4) to the recycling  
7 or disposal site in the same country, again along a default distance of 1000 km.

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**Table 11.6.** Modelling of source-separated collection and transport of plastic-containing WEEE for recycling in the EU <sup>(1)</sup>.

Collection type	Share (%)	Distance (km/t)	Vehicle	Share (%)	Dataset	Amount (km*t/t <sub>collected waste</sub> )
Direct delivery to collection platforms	84.6%	4.2	Van	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	3.55
On-demand collection via public service	15.4%	20	Van	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	3.08
<b>Transport to pre-treatment plants</b>						
Transport	100%	60	Small truck	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	60

<sup>(1)</sup> Based on Biganzoli et al. (2015).2  
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**Table 11.7** Modelling of plastic-containing WEEE collection and transport as residual waste to incineration or landfilling <sup>(1)</sup>.

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

<sup>(1)</sup> Based on Rigamonti et al. (2013).

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#### **11.4.5.3 Modelling of pre-treatment and recycling in Europe**

Separately collected, plastic-containing WEEE are firstly pre-treated in specific facilities, where they are either manually sorted, dismantled and then shredded, or directly shredded and sieved in a fully automated process without any previous dismantling (Jonkers et al., 2016). In the first case, the aim of manual sorting and dismantling is to remove mercury-containing light sources, and other problematic materials or unknown items not intended for continued processes. Fragmented scraps from both processes are then sorted to remove metals (operation not considered in this inventory as not relevant for WEEE plastic components), and to separate bromine containing plastics from the remaining, non-bromine containing fraction through flotation (Jonkers et al., 2016). The removed brominated plastic fraction is typically sent to incineration as hazardous waste. Conversely, the residual plastic-rich fraction (containing up to 90% of plastic material, and normally referred to as Electronics Shredder Residue - ESR) is the feedstock for the subsequent recycling process via further sorting into specific polymer families, removal of remaining contaminants, and regranulation (Schwesig and Riise, 2016).

In this study, 23% of panels in separately collected printers were assumed to follow the partly manual pre-treatment route, with the remaining 77% following the fully automated route. This scenario is in line with the one assumed at the global scale in the *ecoinvent* dataset “[GLO] market for used printer, laser”, which was assumed to extend also to the European situation, in the absence of more representative data on the shares of the two pre-treatment routes in this region.

Inventories related to both types of pre-treatment processes, and for the subsequent sorting process, were developed based on the data reported in Jonkers et al. (2016), which mostly refer to full-scale facilities or large-scale pilot plants operated by a major Swedish recycling company (Stena Metall). Only processes relevant to the WEEE plastic fraction were taken into account in the inventory, while excluding those exclusively applied to other material flows (e.g. metal separation). Relevant activities only involve process-specific burdens, which in the considered data source are only reported per unit mass of total waste entering the pre-treatment process, and were thus implemented as such in the inventory. This means that pre-treatment burdens were equally partitioned among all the components in the processed WEEE, regardless of the respective material (which is equivalent to perform an allocation based on the incoming mass of single material components). Moreover, a 100% processing efficiency was assumed for (non-brominated) plastic parts in WEEE during pre-treatment, thus considering that these are entirely routed to the ESR stream obtained as an output for subsequent recycling. For modelling purposes, process exchanges defined based on the described procedure were combined with EF-compliant datasets representative of EU-average conditions, as summarised in Table 11.8. It is important to note that the developed inventory only partially covers the burdens of the last sorting step to separate bromine-containing from non-bromine plastics, due to absence of data on water consumption for flotation and on the subsequent wastewater release, as well as on the use of any chemicals or detergents. However, the overall energy consumption of the process (i.e. one of the most relevant exchanges for LCA purposes) was included in the inventory.

**Table 11.8.** Life cycle inventory of the pre-treatment process of plastic parts in source-separated WEEE for ESR (Electronics Shredder Residue) production (amounts refer to 1 ton of WEEE plastics for treatment, and are based on Jonkers et al., 2016).

Process/Flow	Share	Flow	Amount	Unit	Dataset	DB	Comments
Material input		WEEE plastic (non-bromine)	1	ton	- <sup>(1)</sup>	-	
Manual dismantling	23%	Electricity	43	kWh	[EU-28+3] Electricity grid mix 1kV-60kV; AC, technology mix   consumption mix, at consumer   1kV - 60kV {34960d4d-af62-43a0-aa76-adc5fcf57246}	EF	Electricity for running conveyor belt, shredding, lighting, heating and ventilation
Shredding (dismantled WEEE)		Electricity	75 <sup>(2)</sup>	kWh			
Automated shredding (non-dismantled WEEE)	77%	Electricity	380	kWh	[EU-28+3] Electricity grid mix 1kV-60kV; AC, technology mix   consumption mix, at consumer   1kV - 60kV {34960d4d-af62-43a0-aa76-adc5fcf57246}	EF	Consumption of a large scale pilot plant for shredding and sieving of possible mercury-containing WEEE
Sorting of bromine-containing plastics (flotation)	100%	Electricity	10	kWh	[EU-28+3] Electricity grid mix 1kV-60kV; AC, technology mix   consumption mix, at consumer   1kV - 60kV {34960d4d-af62-43a0-aa76-adc5fcf57246}	EF	Water consumption, discharge and possible use of chemicals not included
Material output		ESR	1	ton	-	-	

<sup>(1)</sup> A 100% sorting efficiency was assumed during pre-treatment for (non-brominated) plastic parts in WEEE (including printer panels) (as discussed in the main text).

<sup>(2)</sup> Electricity requirement for a shredder that treats a specific pre-sorted WEEE fraction from a specific market.

Mechanical recycling of ABS and PC/ABS from the plastic mixture in the ESR flow recovered from WEEE pre-treatment takes place through a more complex process than recycling of sorted plastics from municipal collection, as already described in Section 11.4.2.2. The process is indeed based on several separation and purification steps, with the purpose of removing remaining contaminants (e.g. residual metals, wood, rubber and foam), separating plastic materials into families with similar densities, and further sorting part of those flows into homogeneous polymer streams. Moreover, compounding with (small amounts of) additives (e.g. modifiers and colorants) may be performed to achieve suitable (mechanical) properties of the recycled material, especially when it needs to be used in more demanding applications. Finally, melt filtration is typically carried out before extrusion, to remove any residual unmelt contaminant (Schwesig and Riise, 2016). A similar process was also assumed to apply to the recycling of PLA and PLA/PC from shredded electronics devices, albeit their recovery from the ESR stream may require additional separation steps.

As discussed in Section 11.4.2.2, no datasets representing the burdens associated with the recycling of specific polymers from (bromine-free) ESR are available. Moreover, a recent LCA study investigating the potential impacts of the process operated by a major recycler of WEEE plastics in Europe (MBA Polymers), reports only partial and not sufficiently detailed information (Wager and Hischier, 2015). Therefore, in the absence of more specific and representative data, an existing EF-compliant dataset related to the production of a generic, secondary plastic granulate out of sorted, post-consumer plastic waste was used as an approximation (*[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*). 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). Process waste and scrap are sent to incineration, consistently with the typical fate of plastic recycling residues, which thanks to their high calorific value are normally routed to incineration or co-combustion in cement kilns (Rigamonti et al., 2014).

The application of this dataset likely underestimates the impacts of the actual recycling process, which deals with a quite heterogeneous feedstock and, as discussed above, includes several separation and purification steps (applying a variety of technologies), while also performing melt filtration before extrusion to remove non-melts (Schwesig and Riise, 2016). Conversely, the modelled process deals with a homogeneous material input (albeit still metal-contaminated, in contrast to ESR), and relies on a simpler layout, including a reduced number of separation steps (i.e. metal separation and a "washing" process to remove small particles, dirt, and residues of oils and fats). Moreover, while additives are reported to be generally used only in some circumstances and in small amounts (Schwesig and Riise, 2016), they were not included in the inventory, due to the lack of reliable, representative and consistent data. Finally, in the specific case of PC/ABS, real process impacts may be further underestimated since the recycling of this polymer is reported to be more challenging than ABS (or other polymers more commonly recovered from the ESR stream such as HI-PS and PP). This is because (among others) the polymer is usually separated into an initial stream of plastics with similar densities, which is not easy to further separate into individual polymer streams. Therefore, additional separation and purification steps are normally required, potentially increasing the energy demand and the potential impacts of the process. This may be the case of the recovery of PLA/PC blends, as well.

Recycled polymer granules of all the investigated panel materials were assumed to replace virgin granules of the same material, whose primary production burdens were credited to the system. Recycled ABS and PC/ABS are reported to achieve similar mechanical properties to the corresponding virgin polymers, possibly through the use of proper additives when the use in more demanding applications is foreseen (Schwesig and Riise, 2016). In this case, a substitution ratio ( $Q_s/Q_p$ ) equal or close to 1 could be applied for such materials, provided that no other relevant material properties are

negatively affected by recycling, and a comparable quality is actually achieved. However, due to incomplete knowledge about the relative quality of the two recycled and virgin polymers, and since additives possibly needed to achieve a suitable quality of the recycled polymers are unknown and could not be modelled in this study, a lower overall quality was assumed for both recycled ABS and PC/ABS compared to their virgin counterpart, according to a conservative approach. In the absence of specific data, a substitution ratio equal to 0.9 was assumed for such polymers, similarly to the default value recommended in the PEF context for other thermoplastic polymers (e.g. PET, PE and PP) when used in packaging applications. The same assumption was then extended also to recycled PLA and PLA/PC blend, again in the absence of more specific data.

To model the burdens of avoided primary production, the same datasets (or combination of datasets) used for the modelling of upstream production of the relevant polymer were applied (as described in Sections 11.4.2, and 11.4.1 for the related feedstock). This was made for consistency reasons, and to avoid possible distortions by applying different datasets from other sources.

According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the pre-treatment, sorting and recycling processes were allocated to the system (assuming a situation of equilibrium between offer and demand of the different recycled polymers, and hence setting the A factor equal to 0.5, similarly to the value recommended in the PEF context for other thermoplastic polymers like PE and PP). Similarly, only 50% of the benefits from avoided virgin material production were assigned to the system itself.

#### **11.4.5.4 Modelling of incineration in Europe**

For all the investigated polymers (ABS, PC and PLA), partially aggregated, material-specific incineration datasets are available from the GaBi database, while no EF incineration inventories are available for these materials. These datasets were thus applied (alone or in combination) to model the End of Life of printer panels in a municipal waste incineration plant.

All the selected datasets are developed based on a waste-specific incineration model, which has been described more in detail in Section 4.4.5.4. 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 general approach to handle energy recovery situations adopted in the PEF context and in the Method (Report I), the product system generating the waste material sent to incineration (i.e. the printer panel life cycle, in this case) takes the full burdens from the incineration process. However, it is credited for 100% of the benefits from avoided production of conventional energy (electricity and heat) replaced by energy recovered from waste. Therefore, the EU residual electricity grid mix (as modelled in the EF dataset "[EU-28+3] Residual grid mix; AC, technology mix | consumption mix, to consumer | 1kV - 60kV") was credited to the amount of recovered electricity, and added to the main process inventory. For recovered heat, a new dataset representing the current EU-average heat supply mix was created, based on background EF 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 includes 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. Also, thermal energy from waste (11%) was excluded, as according to the approach described above for the handling of energy recovery situations, 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).

#### **11.4.5.5 Modelling of landfilling in Europe**

Landfilling of printer panels made of conventional, non-biodegradable polymers (i.e. fossil-based ABS and PC/ABS, both virgin and recycled) was modelled based on a common, aggregated EF dataset representing disposal of plastic waste in a managed, municipal solid waste landfill (*[EU-28+EFTA] Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific sites)*). The same dataset was also applied to model landfilling of PC used as a copolymer in PLA/PC printer panels.

The dataset relies on a material-specific inventory, but it refers to the average chemical composition and degradability values of generic plastic waste, rather than to those of the specific polymer. 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<sup>193</sup>. 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 emissions to the environment over a 100 year time horizon. Emissions occurring beyond 100 years are not accounted for in the model. Landfill gas generation is calculated based on the organic carbon content in the waste and the respective degradation rate assumed over 100 years (not reported), while, 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 rate of landfill gas, energy efficiencies of engines, collection rate of leachate and respective treatment efficiencies). Further details on how these parameters were set to reflect the EU-average situation are provided in Section 4.4.5.5.

For PLA printer panels (and for PLA used as a copolymer in PLA/PC panels), a material-specific landfilling inventory was built, based on the calculation tool developed by Doka (2009b) for the modelling of waste disposal into sanitary landfills. The tool allows to calculate material-specific landfilling inventories accounting for the specific chemical composition and other relevant chemo-physical properties of the landfilled waste (Table 11.9), as well to adjust a number of relevant site- and technology-specific parameters to the reference geography and to the corresponding average landfilling technology. For this purpose, the model applies element-specific transfer coefficients to calculate the distribution of decomposition products originating from elements in the waste composition between landfill gas and leachate, and their ultimate emissions 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.

<sup>193</sup> While more polymer-specific landfilling datasets are available from other databases (i.e. *ecoinvent*), the mentioned EF dataset for landfilling of generic plastic was selected, as specifically referring to EU as the reference geography (in contrast to available polymer-specific datasets), and to keep consistency with the overall dataset selection "hierarchy" followed in this project (Section 3.5).

**Table 11.9.** Elemental composition and lower heating value of PLA considered for End of Life modelling <sup>(1)</sup>.

Element	Share (%)	Element	Share (%)
TS	100	C biogenic (%TS)	49.5
Water	0	H (%TS)	5.60
VS (%TS)	100	O (%TS)	44.5
Ash (%TS)	0	N (%TS)	0.1
C fossil (%TS)	-	S (%TS)	0.3
LHV = 18.4 MJ/kg <sup>(2)</sup>			

<sup>(1)</sup> Based on results from composition analysis related to Ingeo polymer available at: <https://www.natureworksllc.com/What-is-Ingeo/Where-it-Goes/Incineration>

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. carbon to CH<sub>4</sub> and CO<sub>2</sub>) within the landfill. For PLA panels, a 1% biodegradation was assumed, according to the results from Accelerated Landfill Conditions (ALC) tests simulating approximately 100 years of deposition in a biologically active landfill (390 days at 21 °C, Kolstad et al., 2012). These results showed no significant biodegradation of both amorphous and semi-crystalline PLA over such a time horizon, even at the highest humidity level applied for testing (i.e. 65%, which should be considered in a conservative approach). Therefore, the same value of degradability assumed in the Doka (2009b) model for conventional, non-biodegradable polymers such as PET and PE was consistently applied (i.e. 1%). For similar reasons, the same value was also considered for PLA used as a copolymer in panels consisting of PLA/PC, which is a non-biodegradable polymer blend.

Site-specific and technology-specific parameters were set so as to reflect as much as possible the average situation at the EU level, as detailed in Section 4.4.5.5. These include the rates of landfill gas capture, utilisation and flaring, as well as the mean annual precipitation and temperature. For the other parameters (e.g. energy efficiencies of gas engines, mean annual evapotranspiration, landfill height and duration of the filling phase) the default values assumed in the model were kept.

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 database<sup>194</sup>. However, for energy-related flows, background EF datasets were applied.

In line with the time horizon applied for landfill emission modelling, (biogenic) carbon in the landfilled polymer that is not degraded (mineralised) after 100 years from deposition was considered to be never released from the landfill body. As a base case, this missed release of atmospheric carbon taken up during biomass growth is not reflected in Climate Change LCIA results for PLA-based printer panels, since characterisation factors for biogenic CO<sub>2</sub> flows have been set to zero (fully conforming to provisions in the PEF method). However, to show the relevance of this choice on the overall results, the Climate Change impact of both PLA and PLA/PC panels was also calculated by acknowledging the effects of non-released biogenic carbon. For this purpose, a specific CO<sub>2</sub> uptake was modelled in the inventory, expressing the net amount of CO<sub>2</sub> taken up during biomass growth and not released (as CO<sub>2</sub> or CH<sub>4</sub>) during polymer degradation in landfill. Considering a biogenic carbon content in PLA equal to 49.5% (Table 12.9), a

<sup>194</sup> Exceptions are the inputs of diesel (burned in building machine) and pitch, which were replaced with suitable EF datasets, to improve reliability of LCIA results in the Ozone Depletion impact category.

carbon uptake equal to 1.80 kg CO<sub>2</sub>/kg PLA, and 1.16 kg CO<sub>2</sub>/kg PLA/PC blend was calculated, based on the 1% mineralisation rate reported above. This uptake was then characterised with a factor of -1 kg CO<sub>2</sub> eq. per kg CO<sub>2</sub>-C not degraded.

#### **11.4.5.6 Modelling of treatment and disposal outside Europe**

As discussed in Section 11.4.6.1, plastic-containing WEEE exported outside Europe were assumed to be routed to India, where they are either dismantled and recycled in sub-standard conditions (within the informal recycling sector), or improperly disposed via open burning or unsanitary landfilling.

Sub-standard recycling in India was modelled based on a combination of three separate *ecoinvent* datasets, specifically developed in the framework of the Sustainable Recycling Industries (SRI) initiative. Altogether, these datasets represent the burdens associated with the recovery of WEEE plastics in the informal recycling sector in place in such country. A first dataset<sup>195</sup> covers the activities of manual dismantling, sorting by type and clean-up of plastic parts from WEEE. These activities are carried out with minimal or no safety conditions for workers, and mostly through rudimentary techniques and improper disposal methods. For instance, identification and sorting of different plastics is made by noise from breaking and smell of burning, while cleaning and segregation is based on the salt water bath method. The two subsequent datasets<sup>196</sup> separately cover downstream activities of shredding cleaned and sorted plastic parts into smaller flakes, as well as final extrusion of the latter into plastic granules of a specific polymer.

The efficiency of the first dismantling, sorting and cleaning step is equal to 99.6%, while the subsequent shredding into flakes has a 97.4% efficiency. Rejects from both processes are mostly sent to burning in brick kilns (approximated with municipal incineration) and, to a lower extent, to a combination of unsanitary/sanitary landfilling and open burning. The final granulation step has a 100% efficiency, so that the overall processing and recycling efficiency equals 97%.

For implementation in the model, background energy-related datasets were replaced with relevant EF-compliant datasets representative of EU-average conditions, while for the remaining, non-elementary inputs and outputs, default *ecoinvent* datasets were maintained. Moreover, virgin material substitution from recycled polymers was modelled as for End of Life recycling in Europe in terms of substitution (Qs/Qp) ratios and primary production datasets (Section 11.4.6.3). Therefore, the burdens from virgin polymer production in the EU were considered as an approximation for production burdens in extra-EU recycling countries (in this case, India), while a 0.9 substitution ratio was considered for all types of recycled polymers. The overall modelling of recycling and of the resulting virgin material substitution was carried out according to the Circular Footprint Formula, considering a value of the A factor equal to 0.5 (under the assumption of a situation of equilibrium between offer and demand of the different recycled polymers).

Open burning of panels made of all types of polymers was modelled through the same *ecoinvent* dataset, representing the burdens from uncontrolled burning of plastic rejects from electronic plastics recycling in the open environment (*[GLO] treatment of waste plastic, consumer electronics, open burning*). The dataset includes direct, material-specific emissions to air and soil from the open burning practice, assuming no flue gas treatment, and that residues are left on the site where burning takes place. While the inventory is developed considering an average composition of plastic rejects from the recovery waste plastics from post-consumer electronic devices, it was considered a suitable approximation for open burning of all types of polymers, taking into account the small share of waste printer panels following this route (i.e. 7.75%). However, to make

<sup>195</sup> “[IN] Treatment of waste plastic, consumer electronics, manual dismantling, sorting and cleaning, informal sector”.

<sup>196</sup> “[IN] Plastic flake production, consumer electronics, for recycling, by grinding/shredding, informal sector” and “[IN] Plastic granulate production, unspecified, recycled, informal sector”.

the modelling more representative, CO<sub>2</sub> emissions to air were adjusted to reflect the actual, material-specific carbon content in the incinerated polymer. Considering a carbon content equal to 56.9% for ABS, 75.6% for PC, and 49.5% for PLA, the following CO<sub>2</sub> emissions were thus calculated (taking into account the share of the different copolymers in PC/ABS and PLA/PC panels): 2.09 kg fossil CO<sub>2</sub>/kg virgin or recycled ABS to open burning; 2.57 kg fossil CO<sub>2</sub>/kg virgin or recycled PC/ABS blend; 1.82 kg biogenic CO<sub>2</sub>/kg PLA, and 2.15 kg CO<sub>2</sub>/kg PLA/PC blend (1.17 kg biogenic, and 0.98 kg fossil).

Similarly to open burning, a common *ecoinvent* dataset was applied to model unsanitary landfilling of panels made of all types of polymers (*[GLO] treatment of waste plastic, consumer electronics, unsanitary landfill, wet infiltration class (500mm)*). The dataset includes a material-specific inventory for the disposal of average rejects from electronic plastics recycling in an unsanitary, sub-controlled landfill for municipal solid waste. In these partially controlled conditions, all the generated landfill gas is considered to be directly emitted to air (during the first 100 years from deposition), while leachate from landfill is not treated. Emissions to air and groundwater are calculated considering a degradability of the waste over 100 years equal to 1%, which is suitable for all the investigated polymers. These indeed include either non-biodegradable polymers or polymer blends (for which a 1% degradation is typically considered), or biodegradable polymers that have shown no relevant degradation over 100 years of landfilling (i.e. PLA, as better discussed in Section 11.4.6.5). In light of this negligible degradation rate, CO<sub>2</sub> emissions from landfilling are very limited, and no adjustment based on the actual carbon content of the polymer was performed. However, they were totally or partially converted to biogenic emissions when considering landfilling of bio-based polymers (i.e. PLA) or partially bio-based polymer blends (i.e. PLA/PC). Moreover, to calculate a variant of Climate Change results acknowledging the effects of non-released biogenic carbon from landfilling of such bio-based polymers, an uptake was additionally modelled in the inventory, expressing the net amount of CO<sub>2</sub> taken up during biomass growth and not released (as CO<sub>2</sub> or CH<sub>4</sub>) during polymer degradation in landfill. Considering a biogenic carbon content in PLA equal to 49.5% (Table 11.9), an uptake equal to 1.80 kg CO<sub>2</sub>/kg PLA, and 1.16 kg CO<sub>2</sub>/kg PLA/PC blend was calculated, based on the 1% mineralisation rate reported above.

#### **11.4.5.7 Modelling of composting and compost use-on-land**

A waste-specific life cycle inventory was developed for aerobic composting of PLA printer panels, following the approach and the assumptions described in Section 5.4.5.6 (to which the reader is referred for further details beyond those specified below). The modelling was carried out according to the general recommendations specified in the Method (Report I – Sections 4.4.12.4 and 4.4.12.7), while relying on process-specific burdens and element-specific transfer coefficients available in the EASETECH model (Clavreul et al., 2014) for the assumed composting technology (i.e. tunnel composting).

The elemental composition considered in the modelling is the one already reported in Table 11.9, while a biodegradation rate equal to 52% was applied (Table 11.10). The latter was defined as the average of (experimental) values available in the literature for fragments of PLA sheets/bars with a thickness between 1 and 3.2 mm, as well as pot for fragments. Since these values were at least partly determined under real composting or laboratory conditions, they were preferred to the application of the minimum biodegradation rate required by the European standard to evaluate compostability of plastics (EN 14995) i.e. 90%. This has indeed to be proven under optimised laboratory conditions that may not be achieved in real composting facilities (or it may anyway differ from values obtained from real laboratory testing available in the literature).

Compared to biodegradable packaging articles investigated in the former case studies (i.e. food packaging film and food trays), printer panels were assumed to be firstly grinded into smaller pieces (e.g. together with green waste possibly received by the plant) to allow proper degradation to take place during the process (larger articles such as panels are likely unsuitable for direct degradation within composting facilities).

Moreover, consistently with the composting process modelled for the mentioned packaging articles, a further screening step was considered, where 30% of the incoming (grinded) material is assumed to be removed and sent to incineration (modelled as described in Section 11.4.6.4).

Carbon in PLA panels sent to composting was assumed to mineralise according to the 52% biodegradation rate reported above, and the same rate was also assumed for biodegradation of volatile solids (VS). According to the emission factors (transfer coefficients) reported in the EASETECH model, 99.8% of mineralised carbon is converted to CO<sub>2</sub>, the rest (0.2%) being converted to CH<sub>4</sub>. The latter is then mostly oxidised to CO<sub>2</sub> (95%), while only 5% is eventually emitted as Methane, equalling 0.01% of the mineralised carbon. No waste-specific Nitrogen emissions from the composting process were modelled, being the Nitrogen content of PLA negligible (0.1%; Table 11.9).

The amount of compost (biomass and simpler organic compounds) obtained from the biodegradation of PLA panels was calculated as the sum of non-degraded volatile solids, as well as of ash and water<sup>197</sup> originally included in the material sent to composting. Compost was assumed to be entirely applied, by means of tractors, on agricultural land, where biodegradation of the residual material further advance. The emissions from on-land application were estimated based on average emission factors calculated by Bruun et al. (2006), considering a time horizon of 100 years from application, and average factors across all types of temperate soils. During this period, 89.3% of the applied carbon was mineralised to Carbon Dioxide, 0.01% was released as Methane, while the remaining 10.7% was not released (being part of more stable organic compounds)<sup>198</sup>.

The overall composting inventory was implemented in the lifecycle model according to the Circular Footprint Formula, considering it as a case of recycling with a value of the A factor equal to 0.5. Only 50% of the burdens from the composting process were thus allocated to the printer panels life cycle. Conversely, the burdens and benefits from on-land application of compost were entirely allocated to it.

**Table 11.10.** Biodegradability values considered for PLA printer housing panels under aerobic composting conditions.

Polymer	Biodegradation rate (%)	Source
PLA (neat, semi-rigid)	52%	Deconinck and De Wilde (2013) – Grinded bars; 3.2 mm thickness (88%) Hermann et al. (2011) – Film fragments; 1 mm thickness; 1 cm <sup>2</sup> area (55%) UBA (2018) – Pot fragments (13%)

#### **11.4.5.8 Modelling of anaerobic digestion**

Anaerobic digestion of PLA printer panels and subsequent post-composting of the resulting, non-digested or partially digested, residual material were modelled according to the approach and the assumptions described in Section 5.4.5.7 (to which the reader is

<sup>197</sup> It is acknowledged that the water content in the compost output may change compared to the original material sent to composting due to losses and integration during the process. However, the water content of the investigated biodegradable polymers is equal to zero, so that this simplifying assumption does not affect the calculated amount of compost.

<sup>198</sup> As a base case, the effects of non-released biogenic carbon are not reflected in Climate Change LCIA results of PLA printer panels, due to the characterisation factors applied to biogenic CO<sub>2</sub> flows (which were set to zero, fully conforming to provisions in the PEF method). The consequences of this are not significant, since the share of residual, non-emitted carbon is low, equalling 5% of total carbon in the composted polymer. However, to show the relevance of this missed quantification on comparative scenario results, the Climate Change impact of PLA printer panels was also calculated by acknowledging the effects of non-released biogenic carbon. For this purpose, a specific CO<sub>2</sub> uptake was modelled in the composting inventory, corresponding to the amount of non-released biogenic carbon, which is equal to 0.0654 kg CO<sub>2</sub> per kg PLA panel sent to composting.

referred to for further details beyond those specified below). For anaerobic digestion, a waste-specific inventory was developed, based on the general recommendations specified in the Method (Report I - Section 4.4.12.5), while relying on relevant process-specific burdens and element-specific transfer coefficients available in the EASETECH model (Clavreul et al., 2014) for the assumed digestion technology (i.e. wet thermophilic digestion).

The elemental composition considered in the modelling is the one already reported in Table 11.9, while a biodegradation/mineralisation rate equal to 25% was applied, according to the lowest value reported for PLA films in the extensive review conducted by UBA (2018; no rates are available for thicker PLA films/sheets). Since this value refers to a film with a thickness of 0.1 mm, which is lower than the one of PLA panels (2.7 mm), the modelled digestion scenario is optimistic in terms of applied biodegradation rate. However, being such a value derived from real testing (albeit it is not specified whether under laboratory or real plant conditions), it was preferred to the application of the minimum biodegradation rate required by the European standard on compostability of plastics (EN 14995), i.e. 50% when testing anaerobic treatability. This has indeed to be proven under optimised laboratory testing conditions, and may not be achieved in real digestion plants (or it may anyway differ from values obtained from real laboratory testing available in the literature).

Similarly to the process modelled for biodegradable packaging articles investigated in the former case studies, prior to digestion waste PLA panels were assumed to undergo a pre-treatment stage, where they are also shredded to make them suitable for further processing in the digestion plant. In this stage, 30% of the incoming material in panels collected for treatment was assumed to be removed as residue, and sent to incineration (modelled as described in Section 11.4.6.4).

Carbon in PLA panels sent to digestion was assumed to be converted (mineralised) into biogas according to the 25% biodegradation rate reported above. The shares of gasified carbon converted into Methane and Carbon Dioxide were calculated based on the stoichiometry of the anaerobic degradation reaction, taking into account the proportions of C, H, O and N in the digested polymer. Carbon conversion shares are thus material-specific and, in the case of PLA, they are equal to 50% for both Methane and Carbon Dioxide.

The amount of residual, non-digested or partially digested bioplastic material in the digestate output was calculated as the sum of non-degraded volatile solids (VS), as well as of ash and water<sup>199</sup> originally present in the input material to digestion. Residual volatile solids were in turn calculated by assuming a ratio between degraded VS and degraded carbon equal to 1.89 (according to the value applied in the EASETECH model for organic waste).

Aerobic post-composting of such residual material from digestion was modelled considering an open (windrow) composting facility, and the typical average carbon and VS biodegradation (mineralisation) rate achieved for generic organic waste in this type of facilities (i.e. 76.4% according to the EASETECH dataset). Based on transfer coefficients (emission factors) reported in this dataset, 97.76% of mineralised carbon was assumed to be converted to CO<sub>2</sub>, and the remaining 2.24% to CH<sub>4</sub>.

The residual organic material from post-composting of digested panels was assumed to be applied on agricultural land as soil amendment. On-land application was modelled according to the same approach and assumptions briefly described in Section 11.4.6.7 for residual material derived from direct composting of PLA panels (and more extensively detailed in Section 5.4.5.6 for the food packaging film case study).

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<sup>199</sup> It is acknowledged that the water content in the digestate output may change compared to the original material sent to digestion due to losses and integration during the process. However, the water content of the investigated biodegradable polymer (i.e. PLA) is equal to zero, so that this simplifying assumption does not affect the calculated amount of residual material from digestion.

#### 11.4.5.9 Contribution to macro- and micro-plastics generation (including product litter)

The contribution to macro- and micro-plastics loss and release of the analysed printer housing panels scenarios was estimated according to the framework outlined in Annex B of the Method (Report I), and summarised in Section 4.4.5.6 of this report in terms of underlying quantification methods and approaches. In this section, the case-specific parameters considered to apply the latter to the assessed printer housing panels LCA scenarios are reported.

##### 11.4.5.9.1 PLP method

Regarding the estimate of the loss and release of macro-plastics at the End of Life stage (from product littering and waste mismanagement), the product-specific parameters reported in Table 11.11 were considered to apply the first level of the framework ("*PLP method*") to printer housing panels (derived from Peano et al., 2020). Note that for the scenarios based on HDPE release rates are lower due to a higher economic value of the material.

**Table 11.11.** Case-specific parameters of the PLP method applied to the printer housing panels LCA scenarios.

Parameter <sup>(1)</sup>	Value
Littering rate ( $LR_{lit}$ ) (%)	0
Release rate to ocean ( $Rel_{ocean}$ ) (%)	5
Release rate to the terrestrial environment ( $Rel_{terenv}$ ) (%)	95

<sup>(1)</sup> For details on the meaning of each parameter, refer to Annex B of the Method (Report I).

No case-specific parameters are required to calculate the supply-chain loss and release of micro-plastics to ocean and to the terrestrial environment by applying the *PLP method*. Indeed, the contribution of the considered micro-plastics sources (pellet loss and tire abrasion) depends on the magnitude of the relevant life cycle processes (i.e. polymer production and road transport) in the specific printer housing panels scenario (which in turns depend on the reference flow<sup>200</sup> of each scenario). However, the quantification is made by means of default parameters that are not affected by the type of product, polymer or feedstock source (as reported in Annex B of the Method (Report I)).

##### 11.4.5.9.2 Expanded PLP method

The parameters required to calculate the micro-plastics loss and release from marine coatings and road markings are not case-specific, and are reported in Annex B of the Method (Report I). Similar to the other considered micro-plastics sources, the contribution of these additional sources depends on the intensity of relevant (transport) processes in the specific product inventory, by means of the above-mentioned case-unspecific parameters.

As for the calculation of the additional contribution of plastic waste exported for recycling to the loss and release of macro-plastics from waste mismanagement, the case-unspecific parameters described in Annex B of the Method (Report I) were applied, in combination with the EU-average recycling rates estimated for each type of printer housing panels (Section 11.4.5.1), i.e. 31% for all scenarios. Note that for this case study a specific share of the recycling rate taking place outside EU is available and considered for the calculations.

<sup>200</sup> The reference flow is the mass of panel material required to fulfil the functional unit.

#### **11.4.6 Calculation of iLUC impacts**

As a base case, the contribution of Indirect Land Use Change (iLUC) to the potential Climate Change impact of the investigated bio-based or partially bio-based polymers was calculated according to the approach outlined in Section 4.4.16.4 of the Method (Report I). A sensitivity analysis on the applied method has been performed, with the respective result being presented in Section 11.7.7.4.

In order to apply (recalculated) iLUC GHG emission factors from the EU 2015/1513 Directive (EC, 2015), the specific land demand for the production of relevant crops ( $\text{m}^2 \cdot \text{year} / \text{kg crop}$ ) was calculated first. The calculation was based on the aggregated amount of arable and agriculture land occupation flows reported, for the relevant country (e.g. Brazil for sugarcane), in the dataset used to model the production of the specific crop. If the geography of such flows was not specified, all the arable and agricultural land occupation flows reported in the dataset were aggregated. These estimates were checked against values of land demand calculated based on 5-years average crop yields from FAOSTAT (FAO, 2019), which were found to be generally aligned with the former (absolute variation between 3% and 20%). Hence, the values estimated based on land occupation flows were applied, for consistency with the overall LCI modelling of scenarios.

Specific land demand for crop production was then converted into a demand per functional unit, based on the specific consumption of crop for polymer production ( $\text{kg crop/kg polymer}$ , consistently with the values applied in the LCI modelling) and the amount of polymer needed to fulfil the functional unit (reference flow) in the relevant scenario. The iLUC contribution to the Climate Change impact was finally calculated by applying to the latter the recalculated GHG emission factors from the Directive. The described calculation steps to estimate the iLUC contribution to the potential Climate Change impact are summarised in Table 11.12.



**Table 11.12.** Calculation of the iLUC contribution to the potential Climate Change impact of printer housing panels LCA scenarios relying on bio-based polymers.

Scenario / Polymer	Feedstock	Land demand for crop production <sup>(1)</sup> [m <sup>2</sup> *y/kg <sub>crop</sub> ]	Crop demand for polymer production [kg <sub>crop</sub> /kg <sub>polymer</sub> ]	Polymer demand per F.U. [kg <sub>polymer</sub> /FU]	iLUC GHG emission factor [kg CO <sub>2</sub> eq./ (m <sup>2</sup> *y)]	iLUC climate change impact [kg CO <sub>2</sub> eq./FU]
S5 – PLA	Maize (US)	1.06 (0.939)	3.08	3.54	0.0612	0.710
S6 – PLA/PC	Maize (US)	1.06 (0.939)	2.00	3.48	0.0612	0.454

<sup>(1)</sup> Based on arable and agriculture land occupation exchanges reported in the dataset applied to the modelling of the specific crop. Values in parenthesis refer to land demand calculated based on crop yield data from FAOSTAT (5-years average), and are reported as a reference.

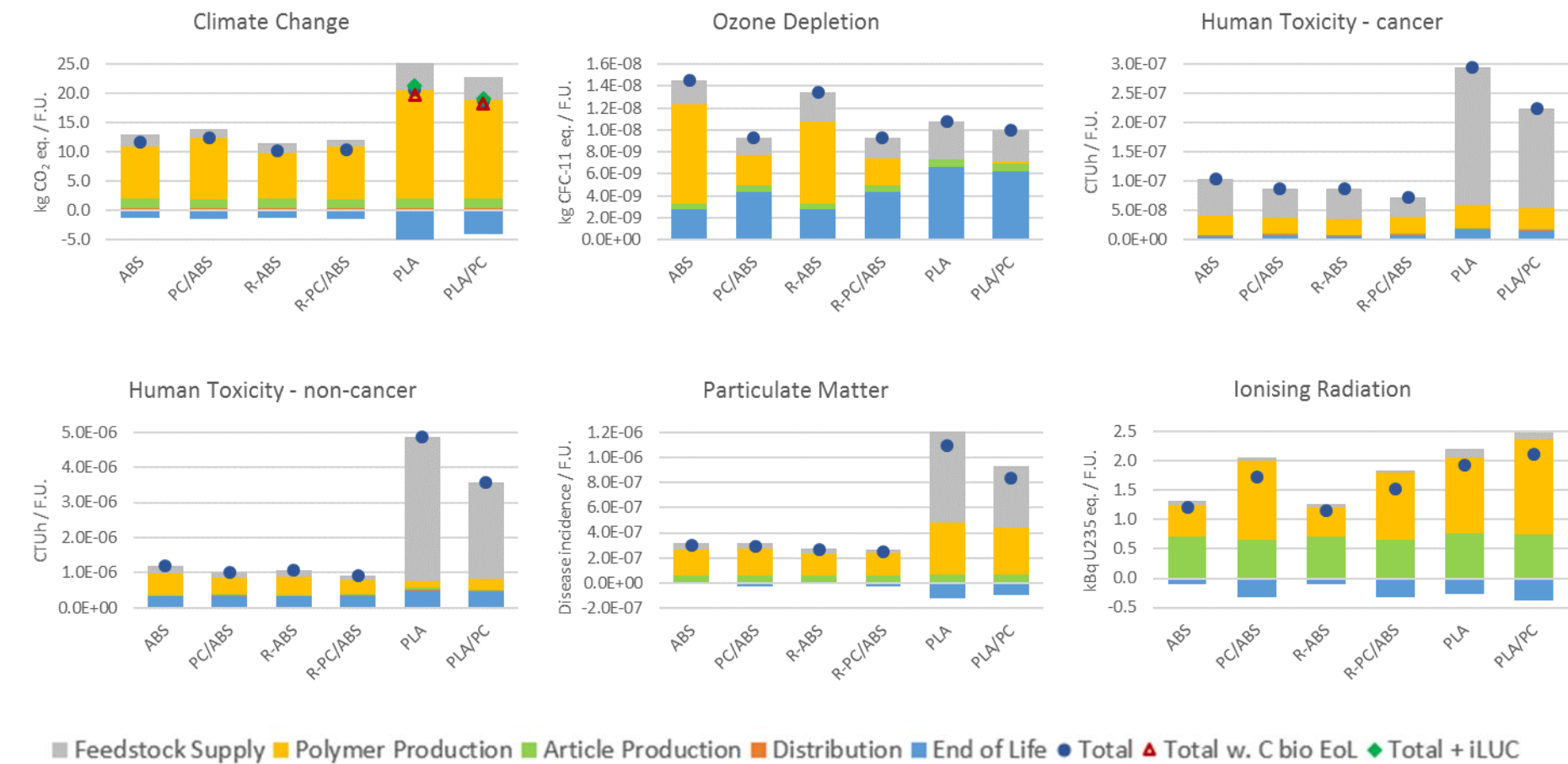
## 11.5 Life Cycle Impact Assessment results

The characterised potential impacts of the examined scenarios (with and without the iLUC contribution to Climate Change) are reported in Figures 11.7 to 11.9. These also show the breakdown of contributions from the main lifecycle stages, which include:

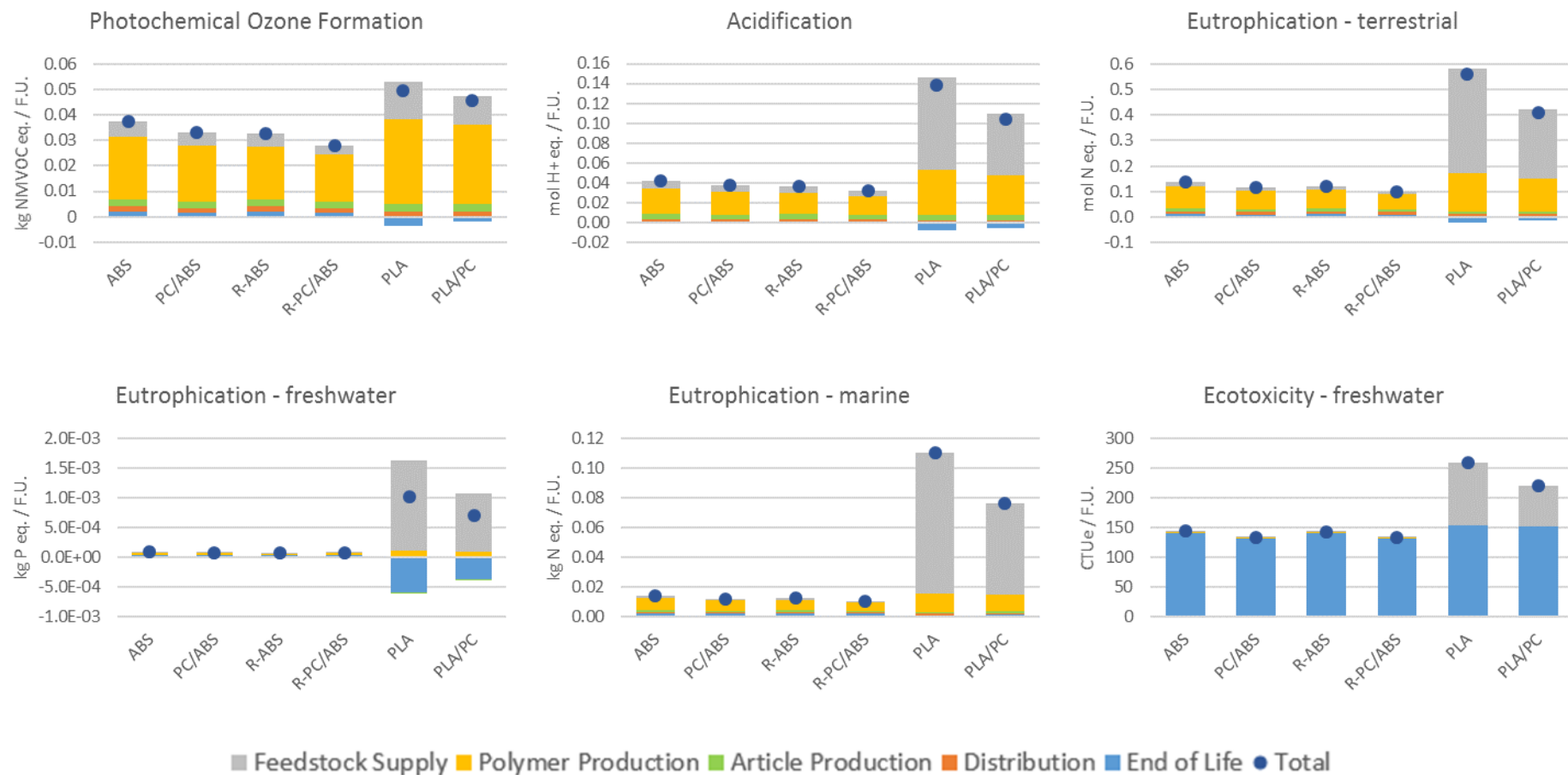
- i) *Feedstock Supply*, oil/natural gas extraction, transport and possible refining, transport of naphtha from refinery to downstream users; collection, transport and pre-treatment of post-consumer plastic in WEEE (recycled polymers); or crop cultivation and transport to further processing (bio-based polymers);
- ii) *Polymer Production*, i.e. all gate-to-gate activities from feedstock processing to monomer/intermediate production and polymerisation, including any transport among these), as well as transport of polymer granulate to the printer panels manufacturing and assembly site;
- iii) *Article Production*, i.e. conversion of the polymer into printer panels by injection moulding;
- iv) *Distribution*, i.e. transport of printer panels from the manufacturing site to the final user, as part of an assembled office printer; and
- v) *End of Life*, i.e. waste printer panels collection, transport and treatment or disposal, as well as any avoided processes from downstream displacement of virgin materials and energy.

Figure 11.10 shows the contribution of each sub-category to the Climate Change impact (i.e. from fossil, biogenic and land use/land transformation GHG emissions), while the overall characterised, normalised and weighted impact assessment results are available in Annex B.8. The latter are not reported in this section to keep readability of the document, despite this is not in alignment with the position suggested in the LCA report template provided in the Method (Report I).

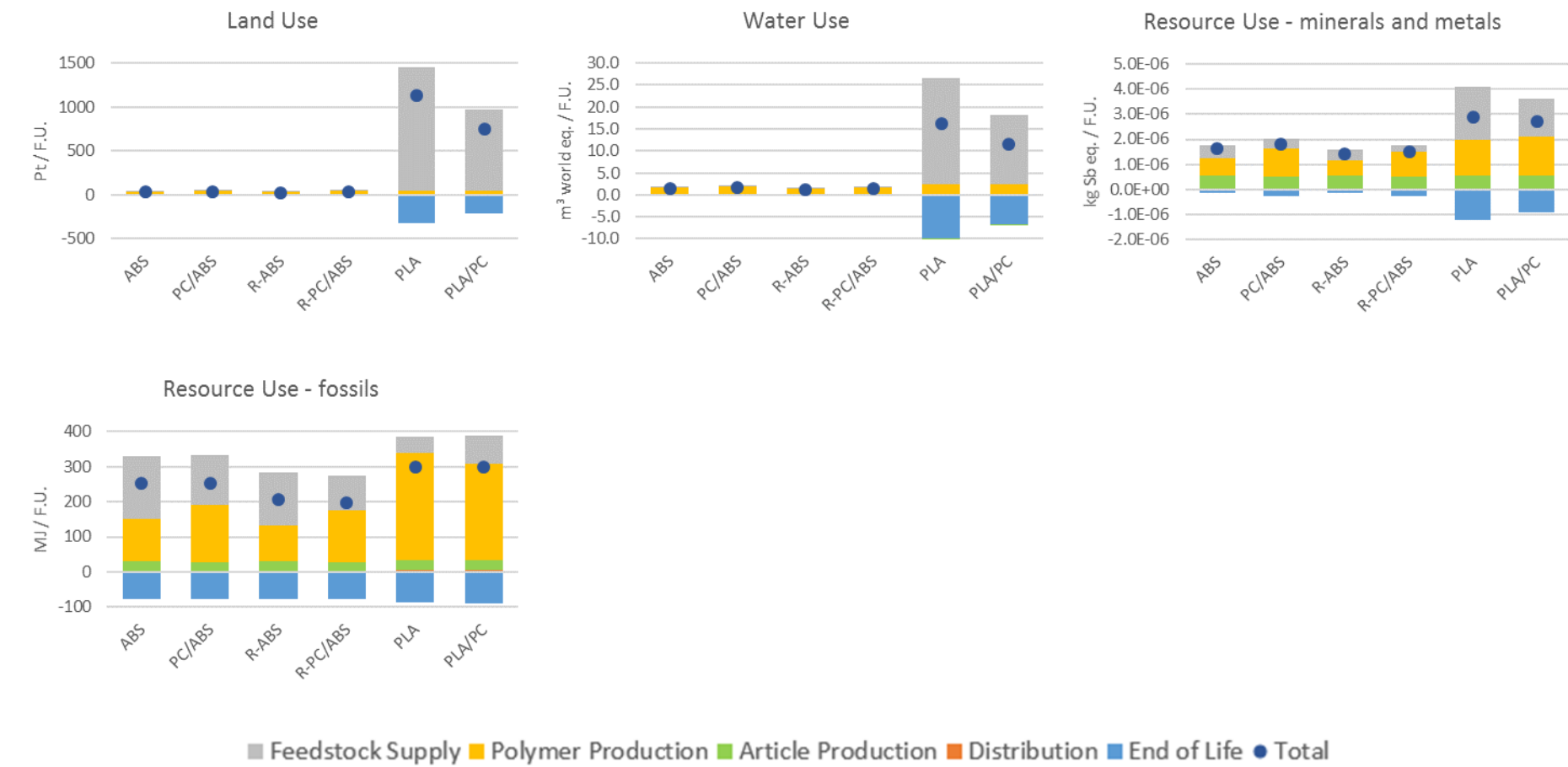
Note that scenario impacts presented in Figures 11.7 to 11.9 refer to the EU-average End of Life scenario (as described in Section 11.4.6.1), and represent the net impacts from waste management activities, resulting from the balance between real burdens and benefits (if any). Potential impacts calculated assuming 100% of post-consumer printer panels being routed to each viable End of Life option are presented in Figures 11.11 to 11.13, except for PLA and PLA/PC panels, which for reasons of scale are separately presented in Annex B.8 (Tables B.8.10 and B.8.11). We warn that this analysis should not be interpreted as a direct LCA comparison of alternative End of Life management options for printer panels, because a product-based perspective is considered, and allocations among consecutive product life cycles are applied to the recycling pathway. Conversely, in a typical waste management LCA that compares alternative End of Life options (i.e. with a functional unit of 1 tonne of waste managed), the recycling pathway would be assigned the full burdens and benefits it involves, as there is no need to break the system mass flows between the system recycling and the one using the recycled feedstock (i.e. system perspective; no allocation needed).



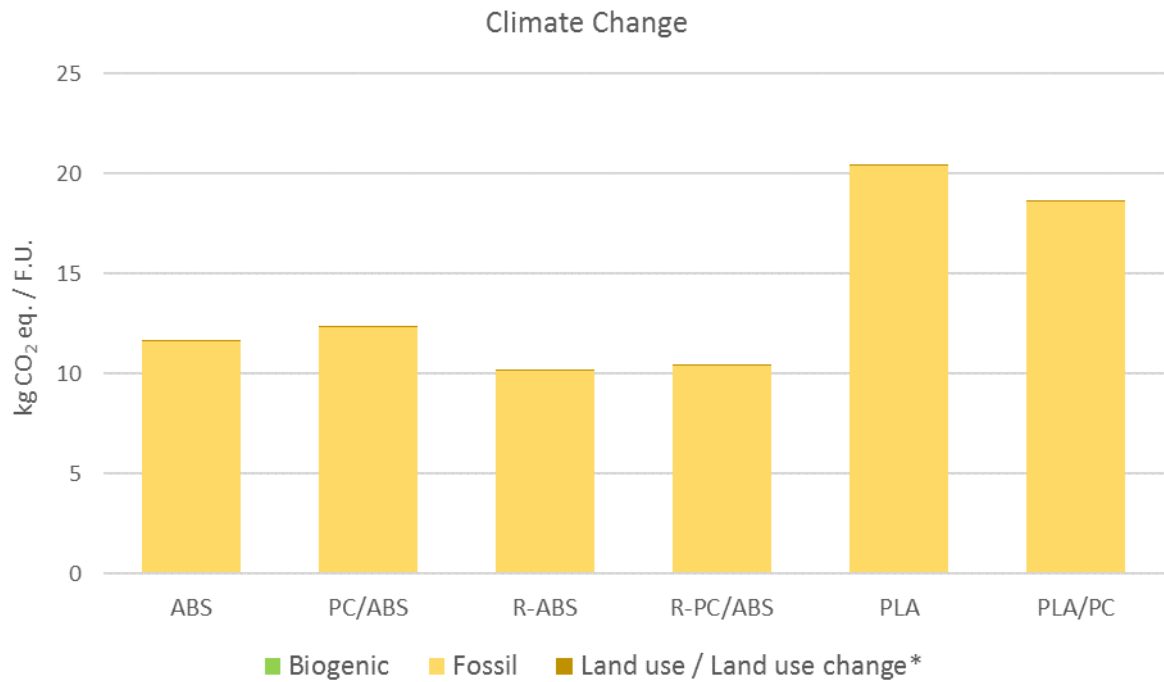
**Figure 11.7.** Potential impact of printer housing panels LCA scenarios for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity – non-cancer, Particulate Matter and Ionising Radiation. Climate Change impacts denoted with “C bio EoL” accounts for the contribution of biogenic carbon not released after 100 years from landfilling of bio-based printer housing panels.



**Figure 11.8.** Potential impact of printer housing panels LCA scenarios for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity - freshwater.

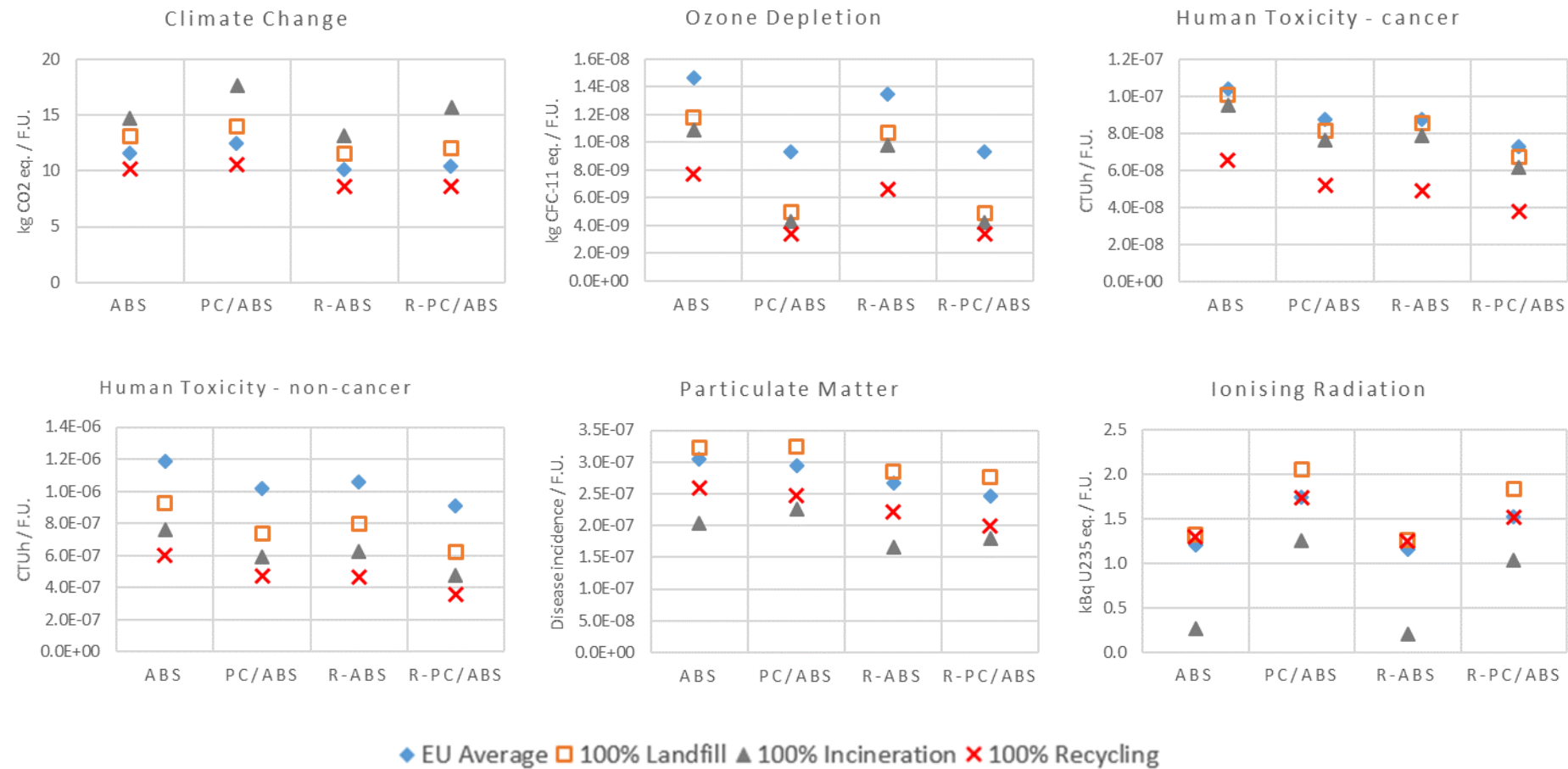


**Figure 11.9.** Potential impact of printer housing panels LCA scenarios for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils.



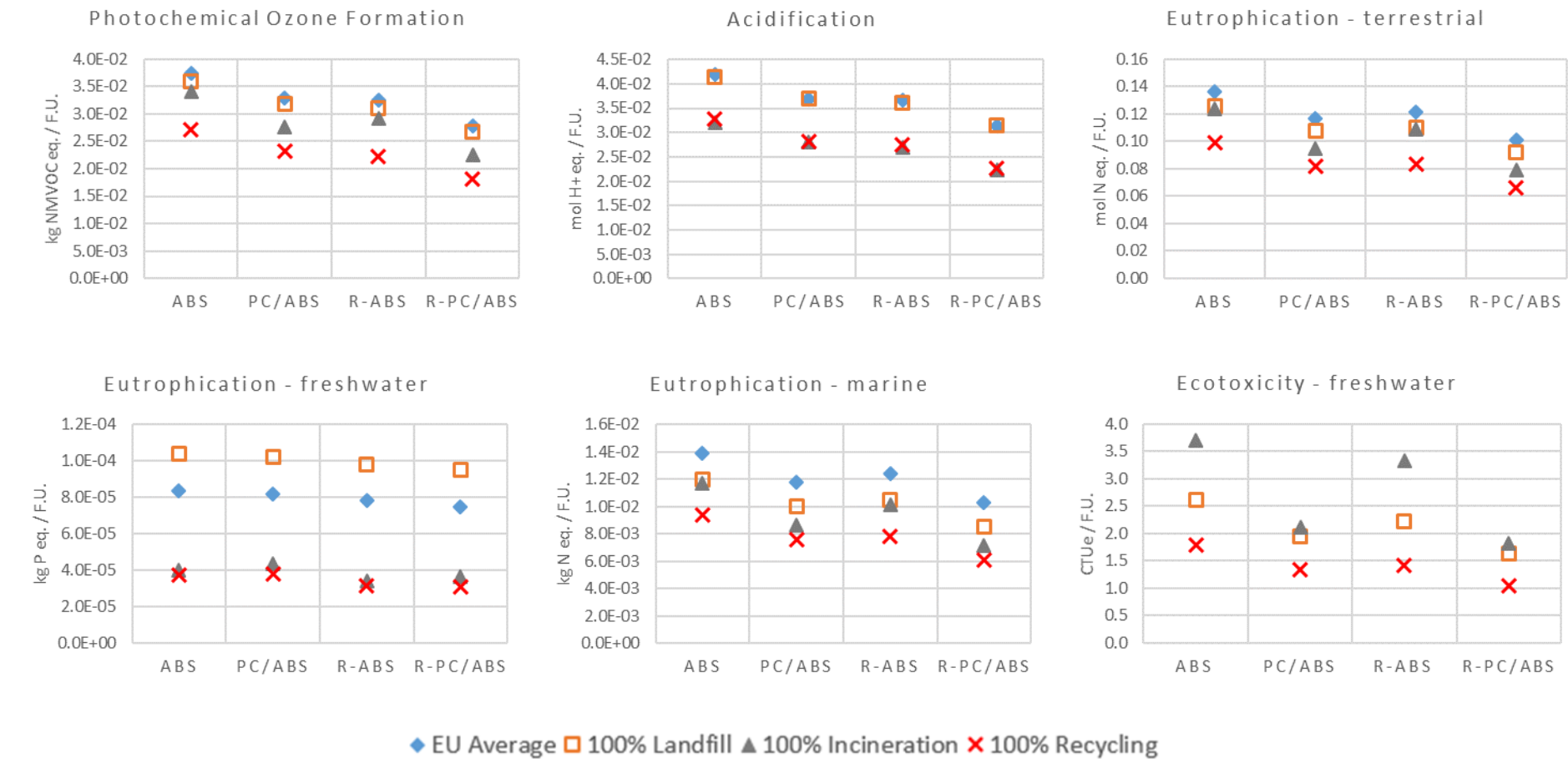
**Figure 11.10.** Contribution of fossil, biogenic, and land use/land transformation GHG emissions to the total Climate Change impact of printer housing panels LCA scenarios (note that the contribution of iLUC to land transformation emissions is excluded).

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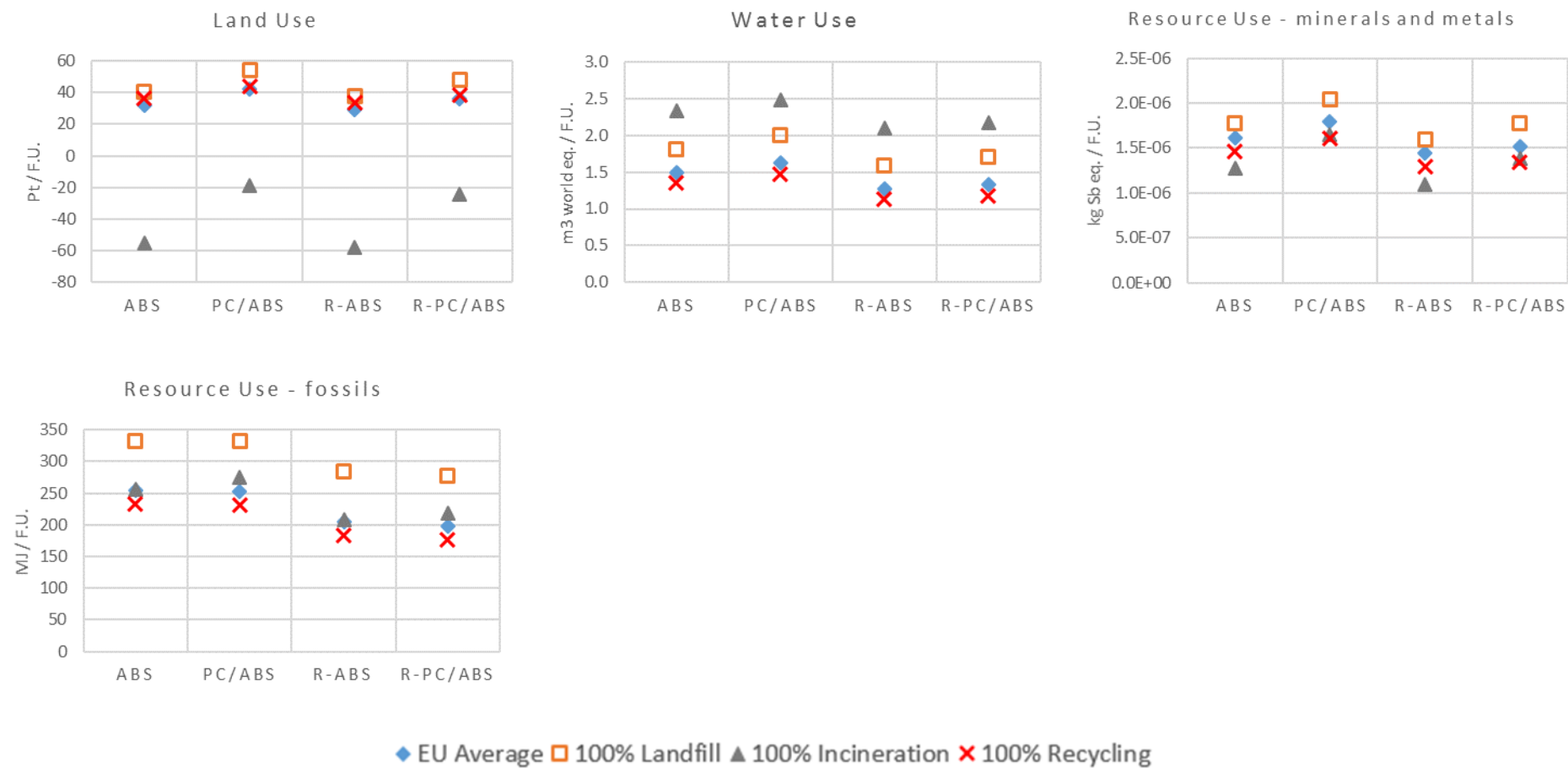
**Figure 11.11.** Potential impact of printer housing panels LCA scenarios for different End of Life options, for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity – non-cancer, Particulate Matter and Ionising Radiation.

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**Figure 11.12.** Potential impact of printer housing panels LCA scenarios for different End of Life options, for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater. Note that the “EU Average” End of Life scenario is excluded from the Ecotoxicity – freshwater results, to better appreciate differences among the other scenarios.





**Figure 11.13.** Potential impact of printer housing panels LCA scenarios for different End of Life options, for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils.

## 11.6 Additional Environmental Information

This section presents additional environmental impacts or aspects going beyond the default impact categories considered in the present method, which reflect official categories adopted in the PEF context. Additional environmental issues addressed include the impact from indirect Land Use Change (iLUC) on Climate Change, potential Biodiversity impacts, as well as the contribution of the investigated printer panels scenarios to macro-plastics formation at End of Life (including product litter) and to micro-plastics generation throughout the supply chain.

### 11.6.1 iLUC impact

Table 11.13 presents the estimated contribution of GHG emissions from iLUC induced by bio-based feedstock supply to the potential Climate Change impact of bio-based printer panels.

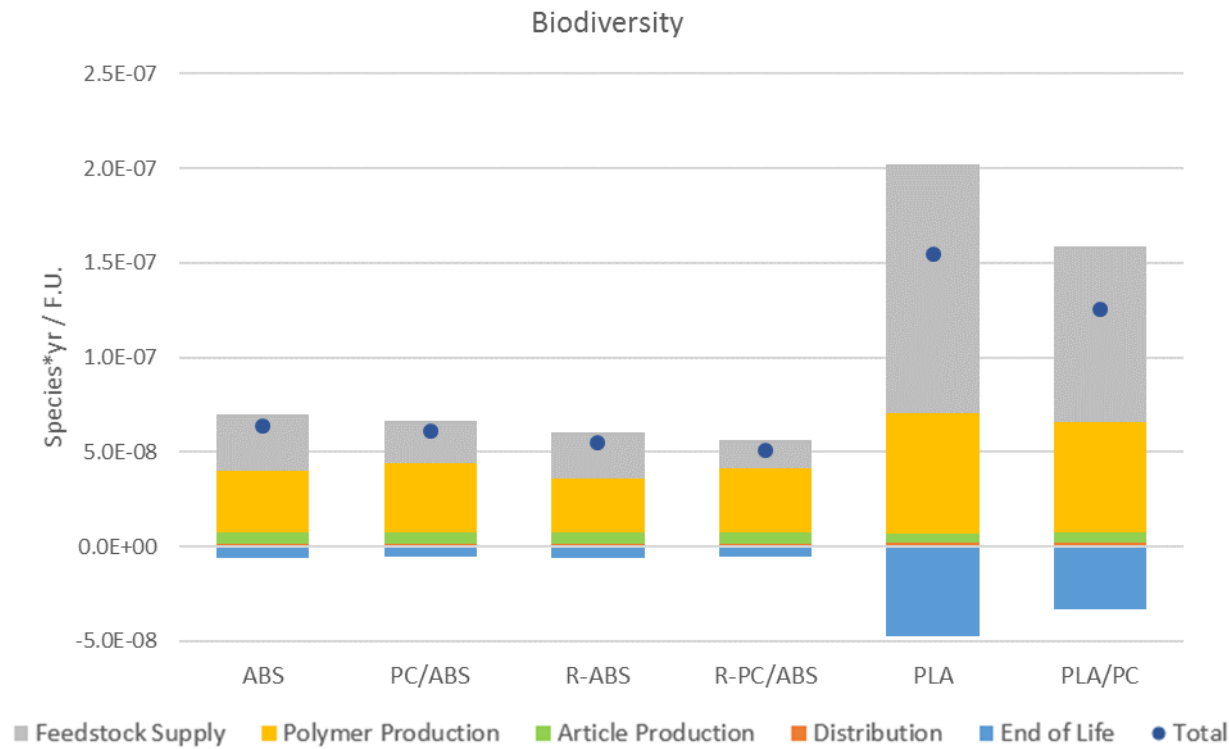
**Table 11.13.** iLUC contribution to the potential Climate Change impact of printer housing panels LCA scenarios.

Scenario / Polymer	iLUC Climate Change impact [kg CO <sub>2</sub> eq./FU]	Total Climate Change impact (incl. iLUC) <sup>(1)</sup> [kg CO <sub>2</sub> eq./FU]
S1 – ABS	-	(11.7)
S2- PC/ABS	-	(12.3)
S3 – R-ABS	-	(10.2)
S4 – R-PC/ABS	-	(10.4)
S5 – PLA	0.71	21.2 (20.5)
S6 – PLA/PC	0.45	19.1 (18.7)

<sup>(1)</sup> Values in parenthesis refer to the Climate Change impact without the iLUC contribution.

### 11.6.2 Biodiversity impact

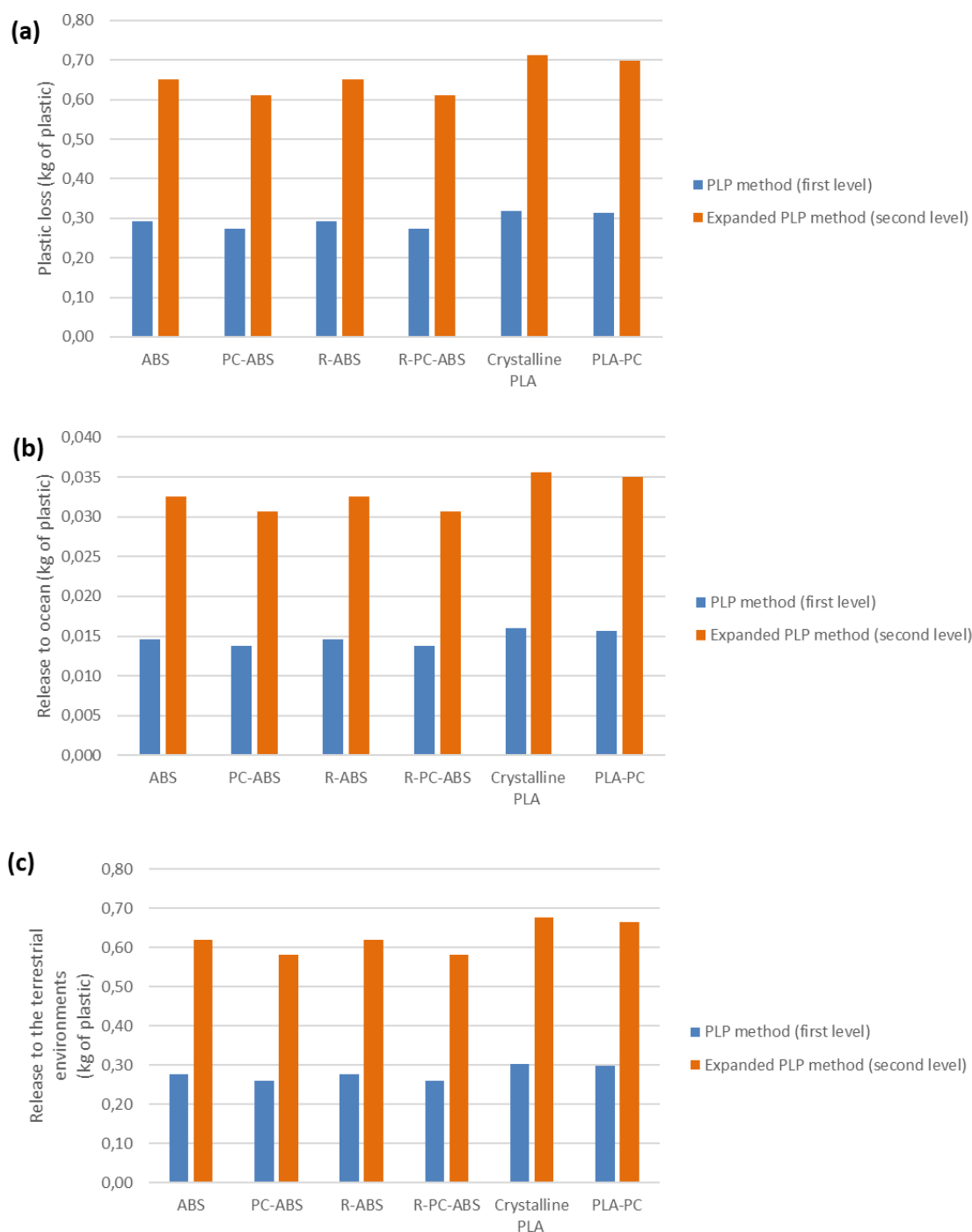
Estimated potential Biodiversity impacts of the investigated scenarios are presented in Figure 11.14. It is reminded that the assessment is performed through an endpoint-level indicator relying on impact assessment methods that for some of the underlying midpoint impact drivers differ from those recommended in the present method.



**Figure 11.14.** Potential biodiversity impact of printer housing panels LCA scenarios, expressed as potential loss of species per year.

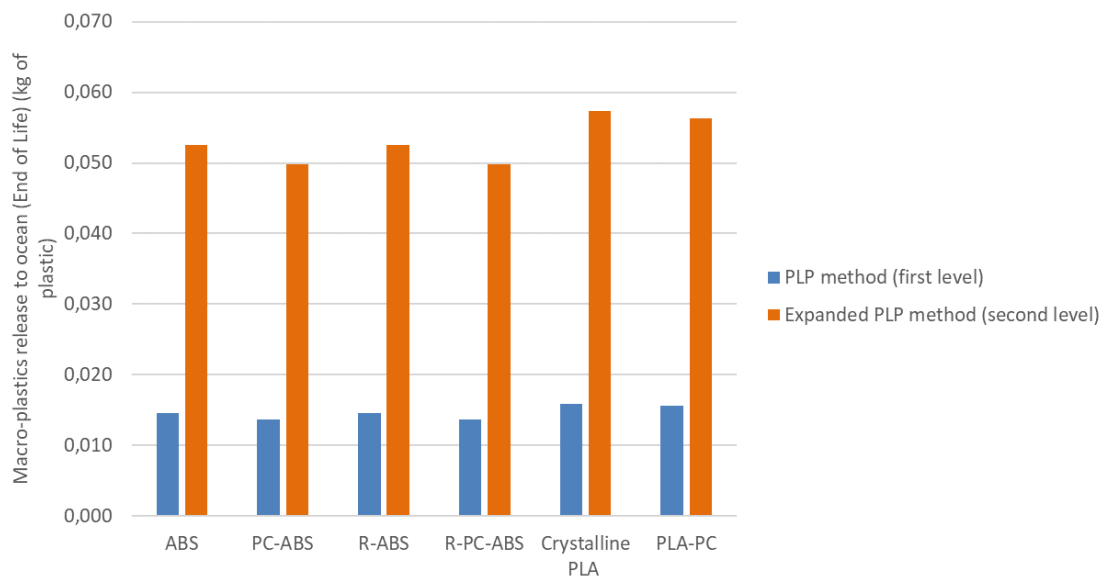
**11.6.3 Macro- and micro-plastics generation**

Figure 11.15 shows the total plastic loss, release to ocean and release to the terrestrial environment estimated for the assessed printer housing panels scenarios, considering the whole product life cycle. Estimates obtained by applying all the two different explored approaches are presented, including the total loss and release to ocean/terrestrial environment calculated through both the *PLP method* (first level) and the *Expanded PLP method* (second level).

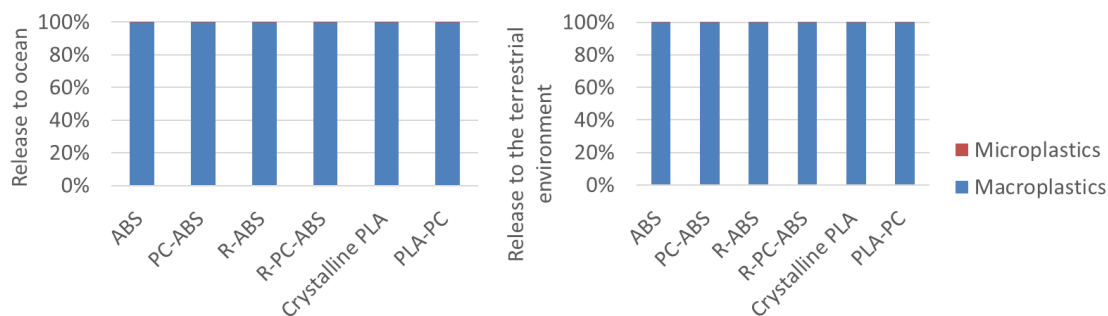


**Figure 11.15.** Total plastic loss (a), release to ocean (b) and release to the terrestrial environment (c) estimated for printer housing panels LCA scenarios with different approaches. The contribution of both macro- and micro-plastics is included, considering the whole product life cycle.

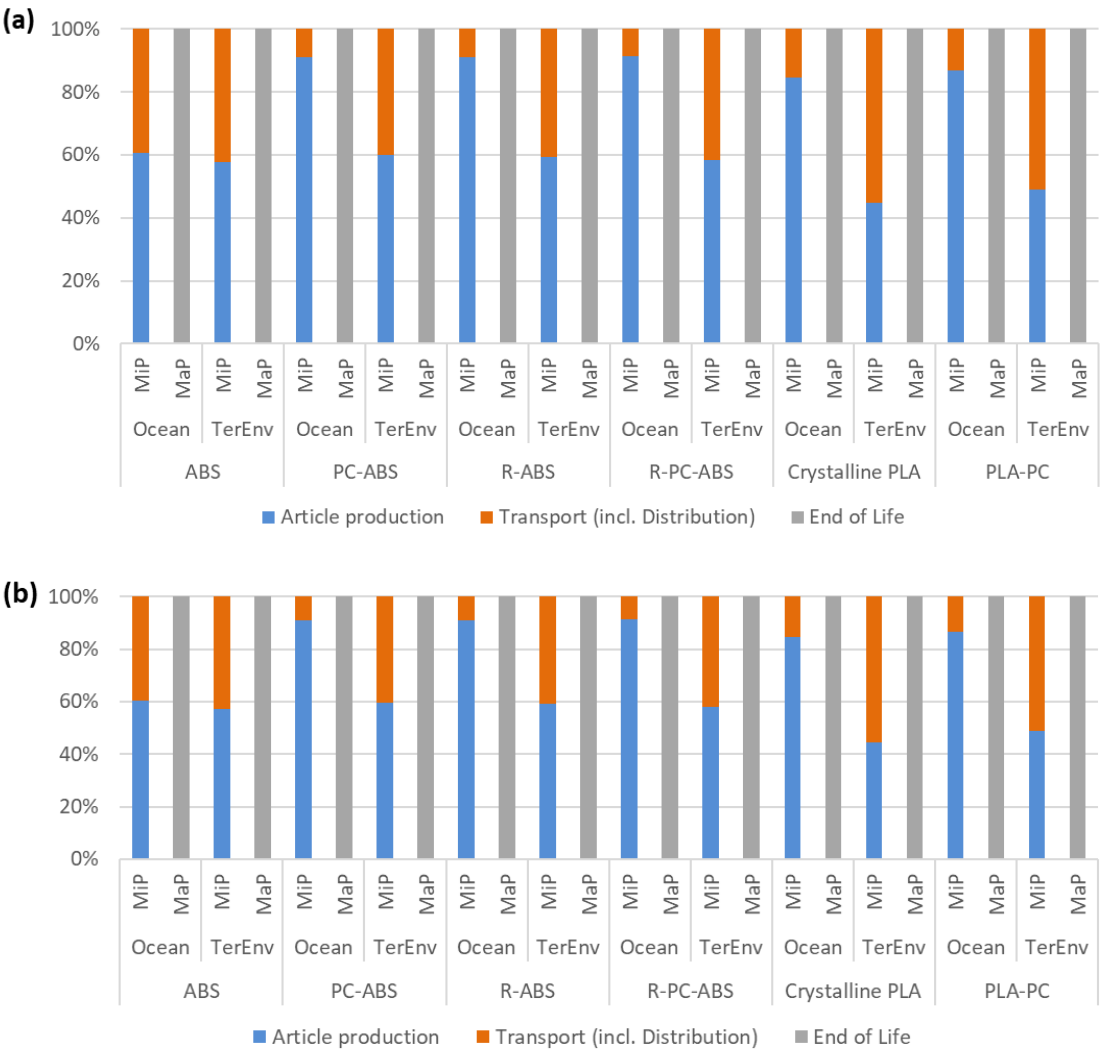
Regarding the sole release of macro-plastics to ocean at End of Life (from product littering and waste mismanagement), Figure 11.16 presents the contribution of each printer housing panels scenario, estimated with the two different explored approaches: the *PLP method* (first level), the *Expanded PLP method* (second level). Note that no littering is considered for this case study.



**Figure 11.16.** Macro-plastics release to ocean at End of Life, estimated with different approaches. As for the share between the release of macro- and micro-plastics, Figure 11.17 shows as an example the relative contribution of the two categories to the total release to ocean and to the terrestrial environment estimated through the *PLP method* (first level). Note that the results are similar to those obtained when applying the *Expanded PLP method* (second level), as further discussed in Section 11.7.7.



**Figure 11.17.** Contribution of macro- and micro-plastics to the total plastic release to ocean and to the terrestrial environment estimated with the *PLP method* for each printer housing panels LCA scenario. The contribution of the different lifecycle stages to the total macro- and micro-plastics release to ocean and to the terrestrial environment of the assessed printer housing panels scenarios has been finally evaluated. The results of this “hotspot analysis” are reported in Figure 11.18 for both the *PLP method* and the *Expanded PLP method*.



**Figure 11.18.** Contribution of life cycle stages to the total macro-plastics (MaP) and micro-plastics (MiP) release to ocean (Ocean) and to the terrestrial environment (TerEnv) estimated for printer housing panels LCA scenarios with (a) the PLP method (first level) and (b) the Expanded PLP method (second level).

## 11.7 Interpretation

In the interpretation of case study results, most relevant impact categories and lifecycle stages of the studied systems are firstly identified (11.7.1 and 11.7.2, respectively). Scenario impacts are then compared, considering both characterised LCIA results (11.7.3), and total normalised and weighted impacts (11.7.4). Results obtained by individually applying each viable End of Life option are separately discussed in Section 11.7.6, while the effects of iLUC and the contribution to macro- and micro-plastics generation are addressed in Sections 11.7.5 and 11.7.7, respectively. Finally, the results of sensitivity analysis on relevant parameters and assumptions are presented (11.7.8).

Most relevant processes were not identified, consistently with the other case studies in this report. However, in this case study the life cycle models of the analysed scenarios present a more similar level of vertical disaggregation of foreground processes within the life cycle (which would allow an at least partly consistent identification of most relevant processes). The identification of most relevant elementary flows was also not undertaken, as this would require prior identification of most relevant processes. Note, however, that any company, organisation or any other supply chain actor applying the

present method shall proceed with the identification of both most relevant processes and elementary flows.

### **11.7.1 Identification of most relevant impact categories**

Table 11.14 shows the most relevant impact categories identified for each printer housing panels scenario, based on normalised and weighted impacts, according to the approach outlined in the Method (Report I – Section 6.2.1). Relevant categories were identified as those that cumulatively contribute to at least 80% of the total normalised and weighted impact of the considered impact categories, excluding toxicity-related ones (these being still based on characterisation factors implemented in the EF 2.0 impact assessment methods, and hence not yet updated based on REACH data<sup>201</sup>). 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, Resource Use – fossils and Acidification are, in this order, the impact categories identified as most relevant for virgin and recycled PC/ABS printer housing panels, as well as for recycled ABS panels. The same categories are also the most relevant for virgin ABS panels, although in this case Resource Use – fossils is the most relevant category, followed by Climate Change. However, both categories provide a similar contribution to the overall normalised impact (36%-40%) in all the mentioned fossil-based scenarios, while the role of Acidification is always marginal (around 5%). The same three impact categories are the most relevant ones also for PLA printer panels (still with a dominant role of Climate Change and Resource Use – fossils), although in this case other categories are also identified as relevant (i.e. Particulate Matter, Water Use and Eutrophication – terrestrial) yet with a lower contribution. Finally, for PLA/PC printer panels, the most relevant impact categories include, again, Climate Change and Resource Use – fossils, followed (with lower shares) by Particulate Matter, Acidification and Eutrophication – terrestrial.

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<sup>201</sup> According to the latest version of the Product Environmental Footprint Category Rules Guidance (EC, 2018), toxicity-related impact indicators based on these characterization factors shall be excluded from the procedure to identify most relevant impact categories.

**Table 11.14.** Most relevant impact categories identified for printer housing panels LCA scenarios and related contribution to the total normalised and weighted scenario impact.

S1 – Fossil-based ABS		S2 – Fossil-based PC/ABS		S3 – Recycled ABS	
Impact category	Contrib.	Impact category	Contrib.	Impact category	Contrib.
Resource Use – fossils	38.5%	Climate Change	38.9%	Climate Change	38.0%
Climate Change	37.1%	Resource Use – fossils	38.1%	Resource Use – fossils	36.5%
Acidification	5.6%	Acidification	4.8%	Acidification	5.7%
<b>Total</b>	<b>81.1%</b>	<b>Total</b>	<b>81.8%</b>	<b>Total</b>	<b>80.3%</b>
S4 – Recycled PC/ABS		S5 – PLA		S6 – PLA/PC	
Impact category	Contrib.	Impact category	Contrib.	Impact category	Contrib.
Climate Change	39.8%	Climate Change	31.3%	Climate Change	33.3%
Resource Use – fossils	36.3%	Resource Use – fossils	21.8%	Resource Use – fossils	25.6%
Acidification	5.0%	Acidification	8.8%	Particulate Matter	7.8%
		Particulate Matter	8.8%	Acidification	7.8%
		Water Use	6.9%	Eutrophication – terrestrial	5.6%
		Eutrophication – terrestrial	6.6%		
<b>Total</b>	<b>81.1%</b>	<b>Total</b>	<b>84.2%</b>	<b>Total</b>	<b>80.2%</b>

### 11.7.2 Identification of most relevant life-cycle stages

Table 11.15 shows the most relevant lifecycle stages in the relevant impact categories identified in Section 11.7.1, and the associated contribution, quantified according to the rules detailed in the Method (Report I – Section 6.2.2). Most relevant stages include those that together contribute to at least 80% of the total characterised lifecycle impact in the specific category, and are highlighted in yellow in Table 11.15. For a more exhaustive picture, the contribution of all lifecycle stages is reported.

Polymer Production is identified as the most relevant stage in the majority of the relevant impact categories identified for both virgin and recycled ABS printer housing panels, followed by Feedstock Supply and, in the case of Acidification, by Article production (although they both provide a more limited contribution). The only exception is represented by Resource Use – fossils, where for both alternatives Feedstock Supply is the most relevant stage, followed by Polymer Production. A similar situation is observed for both virgin and recycled PC/ABS panels, with Polymer Production being the most relevant stage across all the identified most relevant categories and especially in Climate Change, where it is responsible, alone, for more than 85% of the total impact. In Resource Use – fossils, also Feedstock Supply is identified as a relevant stage (contributing to over 50% of the total impact), and the same happens in the case of Acidification for virgin PC/ABS panels (although the contribution is much lower, i.e. 17%). For recycled PC/ABS panels, Article Production is instead the second most relevant



stage in Acidification, followed by Feedstock Supply, both with a contribution around 15%.

Also for PLA and PLA/PC printer panels Polymer Production contributes, alone, to an important share of the total impact in the two most relevant impact categories (Climate Change and Resource Use – fossils), equalling 90% and more. Conversely, for the remaining categories (Acidification, Particulate Matter, Eutrophication – terrestrial, Water Use), Feedstock Supply is the most relevant stage, followed in most cases by Polymer Production, albeit with a more limited contribution (27-44%). The only exception is Water Use (identified as relevant for PLA printer panels), where Feedstock Supply is the only most relevant stage.

**Table 11.15.** Contribution of life cycle stages to the total characterised impacts of the most relevant categories identified for printer housing panels LCA scenarios. Most relevant stages (i.e. those contributing to more than 80% of the total impact) are highlighted in yellow.

<b>S1 – Fossil-based ABS</b>					
<b>Resource Use -fossils</b>		<b>Climate Change</b>		<b>Acidification</b>	
<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	71.0%	Polymer Production	77.2%	Polymer Production	59.9%
Polymer Production	47.4%	Feedstock Supply	17.4%	Feedstock Supply	19.6%
Article Production	10.5%	Article Production	14.1%	Article Production	12.1%
Distribution	1.7%	Distribution	2.7%	Distribution	5.3%
End of Life	-30.6%	End of Life	-11.5%	End of Life	3.1%
<b>S2 – Fossil-based PC/ABS</b>					
<b>Climate Change</b>		<b>Resource Use - fossils</b>		<b>Acidification</b>	
<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	85.1%	Polymer Production	63.9%	Polymer Production	62.8%
Feedstock Supply	12.6%	Feedstock Supply	55.6%	Feedstock Supply	17.3%
Article Production	12.4%	Article Production	9.8%	Article Production	12.9%
Distribution	2.4%	Distribution	1.6%	Distribution	5.6%
End of Life	-12.5%	End of Life	-30.9%	End of Life	1.4%
<b>S3 – Recycled ABS</b>					
<b>Climate Change</b>		<b>Resource Use - fossils</b>		<b>Acidification</b>	
<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	77.1%	Feedstock Supply	72.5%	Polymer Production	57.9%
Feedstock Supply	16.7%	Polymer Production	50.2%	Feedstock Supply	18.7%
Article Production	16.2%	Article Production	13.0%	Article Production	13.8%
Distribution	3.1%	Distribution	2.1%	Distribution	6.0%
End of Life	-13.2%	End of Life	-37.8%	End of Life	3.6%

S4 – Recycled PC/ABS					
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	86.5%	Polymer Production	73.9%	Polymer Production	62.1%
Article Production	14.7%	Feedstock Supply	51.1%	Article Production	15.1%
Feedstock Supply	10.8%	Article Production	12.6%	Feedstock Supply	14.5%
Distribution	2.8%	Distribution	2.0%	Distribution	6.6%
End of Life	-14.8%	End of Life	-39.6%	End of Life	1.6%
S5 – PLA					
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	89.9%	Polymer Production	103.0%	Feedstock Supply	67.5%
Feedstock Supply	24.7%	Feedstock Supply	15.0%	Polymer Production	32.8%
Article Production	8.3%	Article Production	9.7%	Article Production	3.8%
Distribution	1.7%	Distribution	1.6%	Distribution	1.7%
End of Life	-24.6%	End of Life	-29.4%	End of Life	-5.7%
Particulate Matter		Water Use		Eutrophication - terrestrial	
<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	67.7%	Feedstock Supply	146.4%	Feedstock Supply	73.4%
Polymer Production	37.7%	Polymer Production	15.3%	Polymer Production	26.6%
Article Production	4.9%	Distribution	0.1%	Distribution	2.2%
Distribution	1.0%	Article Production	-0.8%	Article Production	1.8%
End of Life	-11.3%	End of Life	-61.0%	End of Life	-4.0%
S6 – PLA/PC					
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	90.0%	Polymer Production	92.5%	Feedstock Supply	59.4%
Feedstock Supply	20.6%	Feedstock Supply	26.7%	Polymer Production	44.2%
Article Production	9.1%	Article Production	9.5%	Article Production	6.4%
Transport	1.8%	Transport	1.5%	Transport	1.3%
End of Life	-21.4%	End of Life	-30.1%	End of Life	-11.3%
Acidification		Eutrophication - terrestrial			
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>		
Feedstock Supply	60.2%	Feedstock Supply	66.2%		
Polymer Production	37.7%	Polymer Production	30.9%		
Article Production	5.0%	Distribution	3.0%		
Distribution	2.3%	Article Production	2.5%		
End of Life	-5.1%	End of Life	-2.7%		

### 11.7.3 Interpretation of characterised results

In this section, the characterised potential impacts of the assessed scenarios (Figures 11.7-11.9) are compared to evaluate the effects of using an alternative feedstock or material for a specific fossil-based polymer of reference used in printer housing panels manufacturing. Therefore, the impacts of partially recycled ABS printer panels are compared with those of virgin ABS panels, while partially recycled PC/ABS panels are compared with virgin PC/ABS panels. Crystalline PLA is mainly intended as a potential replacement for ABS (due to its mechanical properties), so that PLA printer panels are compared with virgin ABS panels. Similarly, PLA/PC printer panels are compared with virgin PC/ABS panels. The impacts of reference scenarios (ABS and PC/ABS printer panels) are also initially compared, to provide useful elements to potentially extend the comparative considerations provided below to any of such scenarios.

The comparison focuses on the sixteen, default impact categories considered in the Product Environmental Footprint (PEF) context and in this assessment. Potential impacts on Biodiversity are discussed separately, as being estimated via an endpoint indicator that is not recommended in the PEF framework, and partially relying on different impact assessment methods for the underlying midpoint-level impact categories. In the following comparison, differences between scenario impacts lower than 10% were not considered significant, in light of the uncertainty associated with the developed life cycle models and the applied impact assessment models. Moreover, any comparative considerations related to Human Toxicity and Ecotoxicity impact categories need to be interpreted in light of the higher uncertainty of the underlying impact assessment models and results. The same applies, to a lower extent, also to Water Use and Land Use. Finally, it has to be reminded that the following comparative considerations refer to a functional unit requiring the achievement of a comparable bending strength performance (and thus reflecting the flexural strength of each material as relevant technical property).

Focusing on the relative performance of printer panels made of the two fossil-based reference materials (i.e. ABS and PC/ABS), PC/ABS printer panels (which has a slightly lower mass compared to ABS panels) show a moderately better performance in seven impact categories (i.e. Acidification, Human Toxicity – cancer, Eutrophication – marine, Eutrophication – terrestrial, Human Toxicity – non-cancer, Ozone Depletion, Photochemical Ozone Formation). In these categories, the impact is decreased between 12% and 36% compared to ABS panels. However, an impact increase is observed in terms of Ionising Radiation and Land Use (+44% and +33%, respectively), while in the remaining categories the two alternatives are comparable.

The use of partially recycled material (35%) for ABS printer panels manufacturing allows a performance improvement in the majority of the impact categories (11 over 16) compared to the use of virgin ABS only. In the remaining categories (Ecotoxicity – freshwater, Eutrophication – freshwater, Ionising Radiation, Land Use and Ozone Depletion) the impacts of both alternatives are comparable. The observed impact reduction is moderate (11-19%), but this also reflects the only partial recycled content assumed for panels. On the other hand, it has to be reminded that the impacts of secondary ABS supply are likely underestimated compared to reality, for the reasons detailed in Section 11.4.2.2. A similar picture of results is observed also for partially recycled PC/ABS printer panels, although impact reductions achieved in this case compared to panels relying completely on virgin material are moderately higher within the single categories, despite falling in a comparable range (11-22%). Moreover, the observed reductions extend to a broader set of categories, since only in three of them the use of the two materials is equivalent (rather than in five). However, also in this case it must be taken into account that the impacts from recycled polymer production are likely underestimated, and to a potentially larger extent compared to ABS panels (the recycling process requires additional separation and purification steps, as better discussed in Section 11.4.2.2)

Crystalline PLA printer panels show substantially increased impacts compared to their fossil-based counterpart (i.e. ABS panels), despite the only moderately higher mass of

material required per functional unit (+9%). The only exception is Ozone Depletion, where the impact is decreased (Polymer Production impacts are significantly higher for ABS printer panels, compared to PLA panels). Moreover, in a few categories, the increase is more limited, i.e. Resource Use – fossils (17%) and Photochemical Ozone Formation (33%). The main responsible for the overall worsened performance is the increased impact from Feedstock Supply, and in some cases of Polymer Production as well. The highest increases are observed for Land Use (35 times, due to an increased land demand for biomass growing), Eutrophication – freshwater (12 times, due to nutrient emissions from cultivation), Water Use (11 times, due to water demand from farming), and Eutrophication – marine (8 times, due again to nutrient emissions). For the remaining categories, the increase is lower, but still significant, ranging between 60% (Ionising Radiation) to 312% (Eutrophication – terrestrial).

In line with the results discussed above, replacing ABS with a higher share of PLA as a blending copolymer with PC increases the impacts of the printer panels life cycle in all the assessed categories except for Ozone Depletion (where the use of both polymer blends is comparable). Again, the highest impact increases are observed in the categories more heavily affected by biomass production (i.e. Land Use, Eutrophication – freshwater, Water Use and Eutrophication – marine), with more moderate increases taking place for Resource Use – fossils, Ionising Radiations and Photochemical Ozone Formation. However, in this case, also the moderately increased mass of PLA/PC printer panels required to achieve a comparable bending strength performance (+14% compared to PC/ABS panels) contributes to this picture of results.

Focusing on estimated potential impacts on biodiversity (Figure 11.14), the two fossil-based reference scenarios (virgin ABS and PC/ABS printer panels) do not show any significant difference (both relies on similar supply chains and related impact drivers). Similarly, partially recycled ABS and PC/ABS panels are essentially aligned with their virgin counterpart, showing only a limited reduction in the range of 14-16%. This reflects the lower burdens from Feedstock Supply and Polymer Production (which are less resource- and emission-intensive), rather than actual differences in supply chain drivers. Conversely, bio-based and partially bio-based printer panels (i.e. PLA and PLA/PC panels) involve a higher impact compared to the respective fossil-based alternative (i.e. +142% and +107%, respectively). This is mostly a consequence of the increased land occupation impacts from feedstock (maize) production, and, to a lower extent, of the increased Climate Change (endpoint) impact from Polymer Production. It has to be reminded, however, that direct potential biodiversity (ecosystem) impacts from oil leakage are not quantified for printer panels made of fossil-based polymers considered as a reference (although emissions from leakage per unit of oil supplied are reported to be quite small; see Section 4.4.1.1).

As a last comment, it is noted how the Climate Change impact of bio-based printer panels (i.e. PLA and PLA/PC panels) is only slightly reduced if the contribution of non-released biogenic carbon after 100 years from landfilling is taken into account. Indeed, a decrease equal to only 4% for PLA panels and 2.5% for PLA/PC panels is observed. This is partly explained by the modest share of landfilling in the applied EU-average scenario (nearly 13% if also unsanitary landfilling outside EU is included) and, in the case of PLA/PC panels, by the only partial biogenic carbon content in the polymer (32%).

#### **11.7.4 Interpretation of normalised and weighted results**

This section briefly compares the investigated scenarios based on the respective total impact scores, calculated by aggregating normalised and weighted impact assessment results across all impact categories (Table B.8.9 in Annex B.8). The comparison is carried out according to the same criteria outlined in Section 11.7.3 in terms of contrasted printer panels scenarios, and considering significant only impact differences larger than 10%. While affected by greater uncertainty (from the calculation and application of normalisation and weighting factors), total impact scores provides a more immediate and synthetic indication of the overall environmental performance of the analysed product

scenarios. This facilitates the interpretation compared to looking at characterised results from several impact categories. However, such estimates are inevitably affected by values choices underlying the definition of weighting factors aimed at establishing an order of relevance for the different impact categories in a European decision context.

The comparison among total normalised and weighted scenario impacts reflects, with some deviations, the overall picture of results emerged when focusing on characterised impact assessment results in single impact categories, and can be summarised in the following points:

- Printer panels made of the two fossil-based reference materials (ABS and PC/ABS) show a comparable overall performance, reflecting the results obtained in the two most relevant impact categories (Climate Change and Resource Use – fossils) and the comparable mass of material required per functional unit by both alternatives.
- Partially recycled ABS and PC/ABS printer panels show a moderate reduction in the overall impact compared to their virgin counterpart (-15% and -18% respectively) in line with the values observed for the two most relevant categories. This limited decrease needs to be interpreted in light of the only partial recycled content assumed as a base case for panels (35%), and higher reductions are expected by an increased uptake of secondary material. However, the same considerations made in Section 12.7.3 regarding the likely underestimated impacts of recycled ABS and PC/ABS supply are still valid (as better discussed in Section 12.4.2.2).
- PLA printer panels show a worsened overall performance compared to their fossil-based counterpart (i.e. ABS panels; +107%), despite the only moderately higher polymer demand per functional unit (+9%). A similar result is observed also for PLA/PC panels, which in view of a 14% higher mass compared to PC/ABS panels, show an increase in the overall impact equalling 76%.

#### **11.7.5 Effects of indirect land use change (iLUC)**

When the contribution of GHG emissions from iLUC is taken into account, only small changes are identified in the Climate Change impact of bio-based alternatives (i.e. PLA and PLA/PC printer panels) (Figure 11.7 and Table 11.13). A maximum increase of 3.5% is indeed observed for PLA panels, while for PLA/PC panels the increase is equal to only 2.4%, reflecting the only partial bio-based content of this polymer (i.e. 65%).

Overall, accounting for iLUC effects does not affect the relative performance of the assessed scenarios (i.e. their comparison is not affected). However, this outcome needs to be read also in light of the GHG emission factors applied for iLUC in this assessment, which appear to fall in the lower end of the range of values available in the recent literature (see Section 4.4.16.3 of the Method (Report I)). The use of an alternative iLUC model and of the resulting emission factors was explored as a sensitivity analysis (see Section 11.7.7.4).

#### **11.7.6 Alternative End of Life options**

This section discusses characterised scenario results calculated by individually applying each viable End of Life option as an exclusive (100%) End of Life scenario (as presented in Figures 11.11 to 11.13 and Tables B.8.10 and B.8.11 in Annex B.8). The main purpose is to evaluate how the performances of the single printer housing panels scenarios are affected by the change in the applied End of Life option, rather than to mutually compare such scenarios for different End of Life options. However, the results should not be interpreted as a strict comparison among different End of Life alternatives for the studied article, since the evaluation applies a product perspective, and burdens/benefits of selected End of Life options (e.g. recycling) are partitioned between different product life cycles (which prevents from capturing the full implications of having a given waste stream routed to a specific End of Life option). In a waste management system

perspective, where allocation is not needed, the total (system-wise) savings associated with the End of Life pathway “100% recycling” would likely be higher than what presented in this report, where a product perspective is applied.

None of the considered product End of Life scenarios (i.e. mechanical recycling, incineration, landfilling, composting and anaerobic digestion) can be identified as preferable across all the assessed impact categories and printer panels scenarios. Moreover, especially for bio-based panels, in several impact categories no relevant changes occur when the different End of Life options are implemented individually, mostly due to the modest contribution of the End of Life stage to the overall scenario impacts. However, within this possibly limited range of variation, the following considerations can be made.

For all printer panels alternatives, 100% mechanical recycling is the preferable scenario in ten impact categories, including Climate Change, Human Toxicity – cancer, Human Toxicity – non-cancer, Photochemical Ozone Formation, Eutrophication – terrestrial, Eutrophication – freshwater, Eutrophication – marine, Ecotoxicity – freshwater, Water Use and Resource Use – fossils. For both virgin and recycled ABS and PC/ABS printer panels, the recycling scenario is preferable also in Ozone Depletion and Resource Use – minerals and metals (limited to virgin and recycled PC/ABS panels). For bio-based printer panels (i.e. PLA and PLA/PC panels) this is the case of Particulate Matter, Acidification, Land Use and Resource Use – minerals and metals, instead. In the few remaining categories (which depend on the scenario) the recycling scenario has an intermediate performance between the incineration and landfilling scenarios.

Incineration is the most favourable End of Life scenario only in the Ionising Radiation impact category (for all printer panels alternatives), mainly because of the large benefits from avoided energy production. For both virgin and recycled ABS and PC/ABS panels, the incineration scenario is preferable also in the categories of Particulate Matter, Acidification, Land Use, and Resource Use – minerals and metals (limited to virgin and recycled ABS panels). For bio-based (PLA and PLA/PC) printer panels, this is the case of Ozone Depletion only. On the other hand, 100% incineration is the least favourable scenario in terms of Climate Change (for all the alternatives except PLA panels) and Water Use.

In many impact categories, 100% landfilling is the worst End of Life scenario for all or most printer panels alternatives, in line with the priority order outlined in the “Waste Hierarchy”, which sets disposal as the least preferable alternative (EC, 2008). Exceptions are Climate Change, Ecotoxicity – freshwater, and Water Use, where for all or many alternatives other End of Life scenarios show the highest impact, as discussed above. Overall, in none of the assessed impact categories and printer panels scenarios landfilling is preferable. This is still valid even when the contribution of non-released biogenic carbon after 100 years from landfilling is taken into account. However, in this case, the 100% landfilling scenario is no longer the worst one for PLA panels, being second only to recycling. The same applies to PLA/PC panels, although in this case the impact of the landfilling scenario is more largely increased compared to that of the most preferable recycling scenario.

End of Life scenarios individually assessed for (biodegradable) PLA printer panels included also biological treatment options, i.e. composting and anaerobic digestion (both followed by on-land application of the residual organic material from such treatments). Although at present such options would be unlikely applied to PLA parts in electronic devices (see Section 11.4.6.1), it was considered relevant to evaluate how they affect the performance of PLA printer panels compared to applying the other viable End of Life options. The analysis shows that none of the two biological treatment scenarios is preferable, in any of the sixteen impact categories. Conversely, composting is the least preferable scenario in a couple of categories (Ionising Radiation and Eutrophication – terrestrial), while it is comparable with landfilling (i.e. the worst or second worst scenario) in several other impact categories. Similarly, anaerobic digestion is the less preferable scenario in two impact categories (Eutrophication – freshwater and Resource

Use – minerals and metals), while being comparable with other low-ranked End of Life scenarios (i.e. landfilling, composting and/or incineration) for many other categories.

Similarly to landfilling, the climate change impact of the 100% composting and 100% anaerobic digestion scenarios was also calculated by accounting for the contribution of non-released biogenic carbon after 100 years from on-land application of the residual organic material obtained from such treatments. For graphical reasons, these results are not displayed in Figure 11.11. However, no significant changes are observed compared to the base case assessment (which excludes the mentioned contribution), due to the very limited amount of residual, non-emitted carbon (equalling 2-5% of total carbon in the original polymer). On the other hand, in this case anaerobic digestion is (immediately followed by composting) the least preferable scenario in place of landfilling (which benefits from a higher impact reduction thanks to the higher share of non-released carbon, i.e. 99%, and is hence the second most favourable scenario after recycling, as discussed above).

As a last comment, it is important to note how in several impact categories the EU average End of Life scenario shows the highest impact compared to applying individual End of Life scenarios (or it has an impact comparable with that of the worst performing scenario(s)). This is mainly because of the important contribution of substandard treatment and improper disposal of waste panels outside the EU (which was not evaluated as a 100% scenario as not being an official route to be pursued). In particular, waste treatment/disposal outside the EU has a remarkably high impact in Ecotoxicity – freshwater, mainly due to antimony emissions from open burning (although these reflects the default average composition of WEEE plastics considered for modelling purposes, which may not be fully representative of the specific materials investigated in this case study). Also in other impact categories the emissions from open burning are the main responsible for such an important contribution of extra EU treatment/disposal, i.e. Human Toxicity – cancer, Human Toxicity – non cancer, Eutrophication – terrestrial and Eutrophication – marine. Conversely, in the case of Acidification and Photochemical Ozone Formation, transoceanic transport outside Europe provides the most important contribution, while for Ozone Depletion, this is provided by diesel used in sub-standard processing of dismantled panels for plastic flake production.

### **11.7.7 Interpretation of results related to macro- and micro-plastics generation**

This section discusses the results presented in Section 11.6.3 on the estimated potential generation of macro- and micro-plastics of the assessed printer housing panels scenarios. The latter are compared according to the same criteria applied to the interpretation of the potential environmental impacts (Section 11.7.3), initially focusing on the results from the two-level approach applied as a base case. No sensitivity analyses were assessed for this case study.

Regarding the total loss of plastics (macro- and micro-plastics) (Figure 11.15), the two base-case approaches (*PLP method* and *Expanded PLP method*) provide similar results when comparing the different printer housing panels scenarios. In both cases, no significant differences are observed among the different scenarios (<7%). The lowest values are observed for the scenarios of PC-ABS and recycled PC-ABS. This is mostly associated to the fact that all scenarios require a similar mass to fulfil the functional unit which contribute to a similar loss of macro-plastics at the End of Life and limited differences are observed in the supply-chain of the scenarios leading to differences in the loss of micro-plastics related to transportation (e.g. tire abrasion). Note, however, that the applied methods rely on mass-based indicators (which are affected by differences in printer housing panel mass) rather than quantifying the number of items (or plastic fragments/particles) lost or released to the environment. If this was the case, all scenarios would result in the same macro-plastics generation at End of Life when applying both *PLP method* and *Expanded PLP method*, since printer housing panels with a lower mass would not imply a reduction in the contribution of product littering and waste

mismanagement (which are based on rates that are not affected by the type of feedstock or material). In turn, an identical macro-plastics generation at End of Life would imply having a comparable total loss, these being mostly determined by macro-plastics (see below).

Comparing the two base-case approaches, the *Expanded PLP* method results in a 220% higher plastic loss and release to ocean and the terrestrial environment compared to the original *PLP method* (Figure 11.15). This is mainly due to an increased contribution of macro-plastics loss at End of Life, which also accounts for mismanagement of plastic waste exported for (sub-standard) recycling outside the EU (where the overall waste mismanagement share is increased by 12% compared to the assumption that all plastic waste is recycled in Europe). Conversely, the two additional micro-plastics sources considered in this alternative approach (road markings and marine coatings) does not significantly affect the results, due to the overall modest contribution of micro-plastics to the total release (in terms of mass).

Focusing on the relation between total loss and release (which depends on the parameters of the *PLP method*), the plastic loss is modelled as fully released to the environment due to the lack of informal waste collection for this type of product. In relative terms, release to ocean accounts for a marginal share of the plastic loss (5%), while release to the terrestrial environment is the most relevant (95%). As shown in Figure 11.17 for the *PLP method*, the total plastic release to both ocean and the terrestrial environment is dominated by the release of macro-plastics at the End of Life (more than 99%), while micro-plastics released throughout the upstream life cycle have only a minor role. Similar results are also observed for the *Expanded PLP method* (not shown), despite two additional sources of micro-plastics are considered in the latter (i.e. road markings and marine coatings). In both cases, this is a consequence of the prevailing mass of macro-plastics, which is directly related to the mass of printer housing panels required per functional unit (reference flow), by means of higher loss and release rates compared to micro-plastics. Therefore, the mass of macro-plastics is at least one order of magnitude higher than the mass of micro-plastics, which instead depends on the quantity of relevant lifecycle processes by means of (much) lower loss and release coefficients (depending on the source).

With regard to the origin of released plastics (in terms of life cycle stage and source), macro-plastics entirely originate from the End of Life stage (Figure 11.18), as they are only derived from mismanagement of the product as waste. Note that no littering is assumed for printer housing panels as it is considered that printer housing panels are managed following specific End of Life collection and management pathways. On micro-plastics release, the relative contribution of the different sources (e.g. loss of plastic pellets and tire abrasion) depends on the characteristics of the supply-chain (e.g. feedstock origin, which affect the contribution from transport across the life cycle), as well as on the mass of polymer required per functional unit (which affect, to different extents, all the considered sources). However, within the limited role of micro-plastics in the overall release to the environment, the most relevant contribution is generally provided by pellet losses from the Polymer and Article Production stages, while transport-related sources have a more restricted role. Exceptionally, transport emissions are relevant (approximately 55%) for the use of crystalline PLA scenario, whose supply-chain represents larger transport distances.

In conclusion, due to the dominant role of macro-plastics loss and defined parameters for release rates to the environment at the End of Life of printer housing panels, a better overall performance is associated to those scenarios employing polymers with a higher economic value and with the lowest release rates. In this context, the use of fossil-based and recycled PC-ABS printer housing panels are the scenarios with the lowest plastic release to both ocean and the terrestrial environment, although differences are not significant due to the similar mass required to fulfil the functional unit. From a methodological perspective, the different approaches provide similar results when comparing alternative scenarios.



### 11.7.8 Sensitivity analysis

A sensitivity analysis has been performed on a number of relevant parameters or assumptions, to evaluate the effects of their variation on the potential impacts of the affected scenario(s), and on the comparative LCA results. The following aspects have been specifically considered:

1. Alternative reference flow;
2. Recycled content in R-ABS and R-PC/ABS printer panels;
3. Feedstock origin and production country of PLA used in neat PLA and PLA/PC printer panels;
4. iLUC model and factors;
5. Additives in printer housing panels.

The following sections present the outcome of the sensitivity analysis for each of the aspects above. The results are reported by comparing recalculated impacts of the affected scenario(s) with those of the base case assessment, of the corresponding reference scenario(s), and, where relevant, of the alternative feedstock scenarios evaluated for the same reference material. In case of changes in parameters contemporarily affecting the impacts of more scenarios, they are all taken into account in the presentation of the results, along with the respective reference scenarios.

#### 11.7.8.1 Alternative reference flow

As a base case, scenario impacts were calculated based on polymer demands (reference flows) estimated assuming that the prioritised performance requirement (out of those expressed in the functional unit) is the need for the panels of not breaking during use under a minimum specified bending load (i.e. of ensuring a minimum bending/flexural strength, as better discussed in Section 11.2). However, prioritising alternative performance requirements may lead to different reference flows (and thus results) since the compared materials perform differently towards the different technical properties affecting their ability to fulfil such requirements. Therefore, reference flows and the corresponding scenario impacts were recalculated considering the need of ensuring a limited deformation under a minimum specified bending load (i.e. of ensuring a minimum bending/flexural stiffness) as the priority performance requirement to be fulfilled, in order to evaluate the robustness of the results obtained as base case.

The reference flow was calculated by means of Material Substitution factors (MSFs) and Material Indices (MIs), following the approach already applied as a base case and described in Section 11.2. The Material Index reported by Ashby (1999) when stiffness is the priority performance (or limiting design constraint) for flat panels loaded in bending (with thickness as a free variable) is represented by the ratio:  $MI = E^{1/3}/\rho$ , where  $E$  is the flexural modulus and  $\rho$  is the density of the material. This index was thus considered appropriate to calculate MSFs for each alternative panel material and the corresponding reference flow. The main calculations carried out for this purpose, and the values assumed for relevant material properties are summarised in Table 11.16. As in base case calculations, an ABS panel with a thickness of 3 mm was considered as a reference, along with the bending stiffness performance achievable with this configuration based on the relevant technical properties of the material.

**Table 11.16.** Calculation of the reference flow for printer housing panels LCA scenarios when bending stiffness is the prioritised performance requirement to be fulfilled (the calculation is based on Material Substitution Factors).

Polymer	Density - $\rho$ (g/cm <sup>3</sup> ) <sup>(1)</sup>	Flexural modulus - E (kN/mm <sup>2</sup> ) <sup>(1)</sup>	MI ( $E^{1/3}/\rho$ )	MSF (m/m <sub>ref</sub> )	Reference flow (kg/FU)	Thickness (mm) <sup>(2)</sup>
ABS R-ABS	1.05	2.10	1.22	1	3.15 <sup>(3)</sup>	3.00
PC/ABS R-PC/ABS	1.17	2.50	1.16	1.05	3.31	2.83
PLA (high heat, crystalline)	1.25	3.90	1.26	0.969	3.05	2.44
PLA/PC blend	1.26	2.55	1.08	1.12	3.54	2.81

<sup>(1)</sup> Derived from Broeren et al. (2016). Values of flexural modulus are not reported as a function of thickness

<sup>(2)</sup> The thickness of ABS panels (reference material) was assumed (and used to calculate the reference flow, see note 3). For panels made of the other materials, the thickness was backward calculated for checking purposes, based on the estimated reference flow and the assumed panel extension (1 m<sup>2</sup>). The estimated thickness values fall within a reasonable range and are comparable with the thickness of ABS panels assumed as a reference.

<sup>(3)</sup> For ABS panels (reference material), the reference flow is calculated assuming a panel thickness of 3 mm, in line with the reference value considered in Broeren et al. (2016). This thickness could not be specifically related to the assumed flexural modulus, and it is noted as a partial inconsistency.

The results of the sensitivity analysis are shown in Figure 11.19, where scenario impacts are reported as a function of the impacts of virgin ABS printer panels, which are set as a 100% baseline, since the reference flow and the impacts of this scenario do not change compared to the base case assessment. The impacts of recycled ABS printer panels are not presented, these being also not affected by this sensitivity analysis (nor is their comparison with the impacts of virgin ABS). The same applies to the comparison between recycled and virgin PC/ABS panels (both scenarios undergo the same change in the reference flow), but both of them are reported for completeness. Recalculated impact values for each of the analysed printer panels scenarios are also available in Annex B.8 Table B.8.12.

In this updated picture of results, the comparison between the two fossil-based reference scenarios (i.e. PC/ABS and neat ABS printer panels) is negatively affected by the now slightly higher mass of polymer required per functional unit in the case of PC/ABS panels (in contrast to the base case, where a slightly lower amount was required compared to ABS panels). As a consequence, PC/ABS printer panels now outperform ABS panels only in one impact category (Ozone Depletion; -29%), rather than in seven categories as in the base case assessment. In several (nine) impact categories, both alternatives are now comparable, albeit PC/ABS panels show (moderately) higher impacts (+19-61%) in six categories (and not only in three of them as in the base case assessment)<sup>202</sup>.

The comparison between PLA printer panels and their fossil-based counterpart (i.e. ABS panels) is not significantly improved, despite the slightly reduced mass of PLA now required per functional unit compared to ABS (in contrast to the base case assessment, where the PLA demand was moderately higher). Indeed, PLA printer panels still show a higher impact across all impact categories except for Ozone Depletion (where as in the base case the impact is lower) and also Resource Use – fossils (where both alternatives are now comparable).

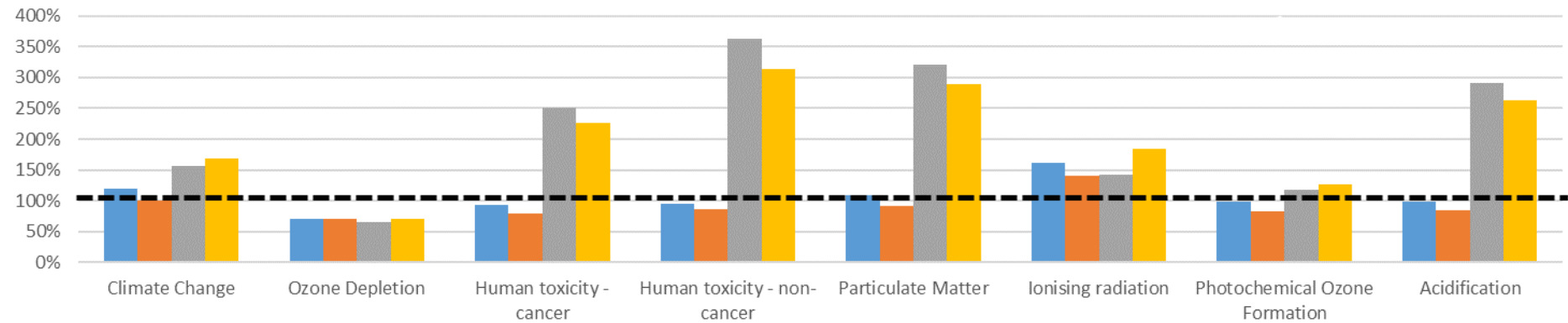
A similar situation is also observed in the comparison between PLA/PC printer panels and their fossil-based reference (i.e. PC/ABS panels) since the relative difference between the mass of the two materials required per functional unit is not significantly affected

<sup>202</sup> These include Ionising Radiation (+61%), Land Use (+49%), Resource Use – minerals and metals (+23%), Water Use (+22%), Climate Change (+19%), and Resource Use – fossils (+12%).

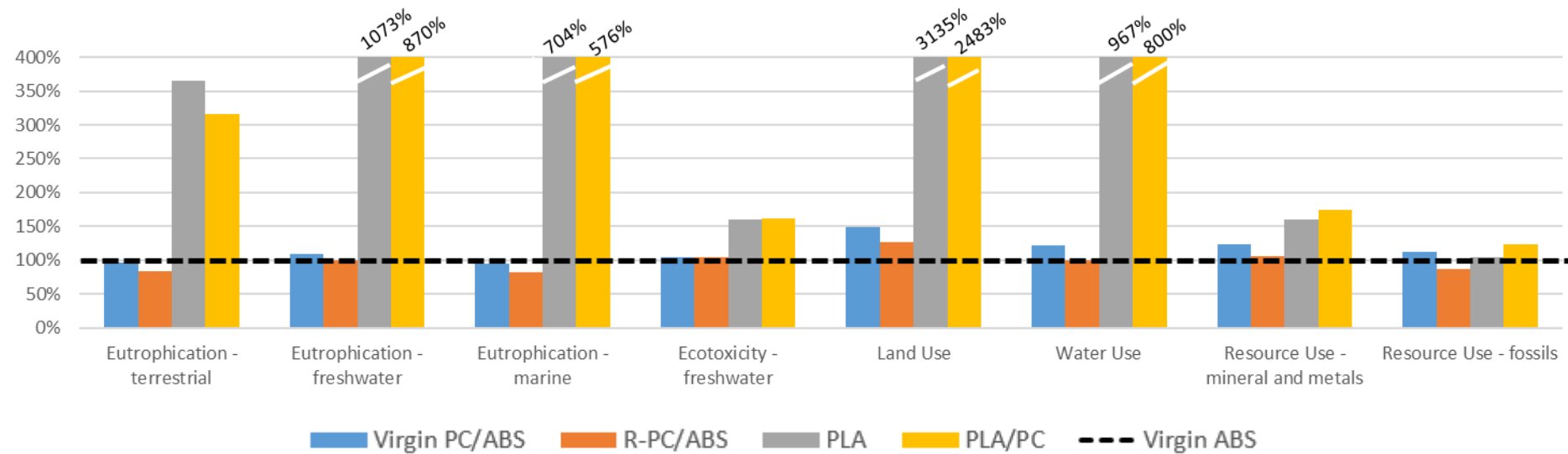
1 compared to the base case assessment (with PLA/PC still being required in a slightly  
2 higher mass than PC/ABS to achieve a comparable performance). Therefore, PLA/PC  
3 panels still show a worse performance across all the investigated impact categories  
4 except for Ozone Depletion, where they are comparable with PC/ABS panels.

5

1



2



3

**Figure 11.19.** Results of the sensitivity analysis on the prioritised performance requirement to calculate the reference flow of printer housing panels LCA scenarios (i.e. ensuring a minimum bending stiffness). Scenario impacts are expressed as a percentage of the impacts of virgin ABS printer panels, which are not affected by the explored change and are set as a baseline. Note that in some impact categories a part of the results is out of scale and is curtailed.

### 11.7.8.2 Recycled content in R-ABS and R-PC/ABS panels

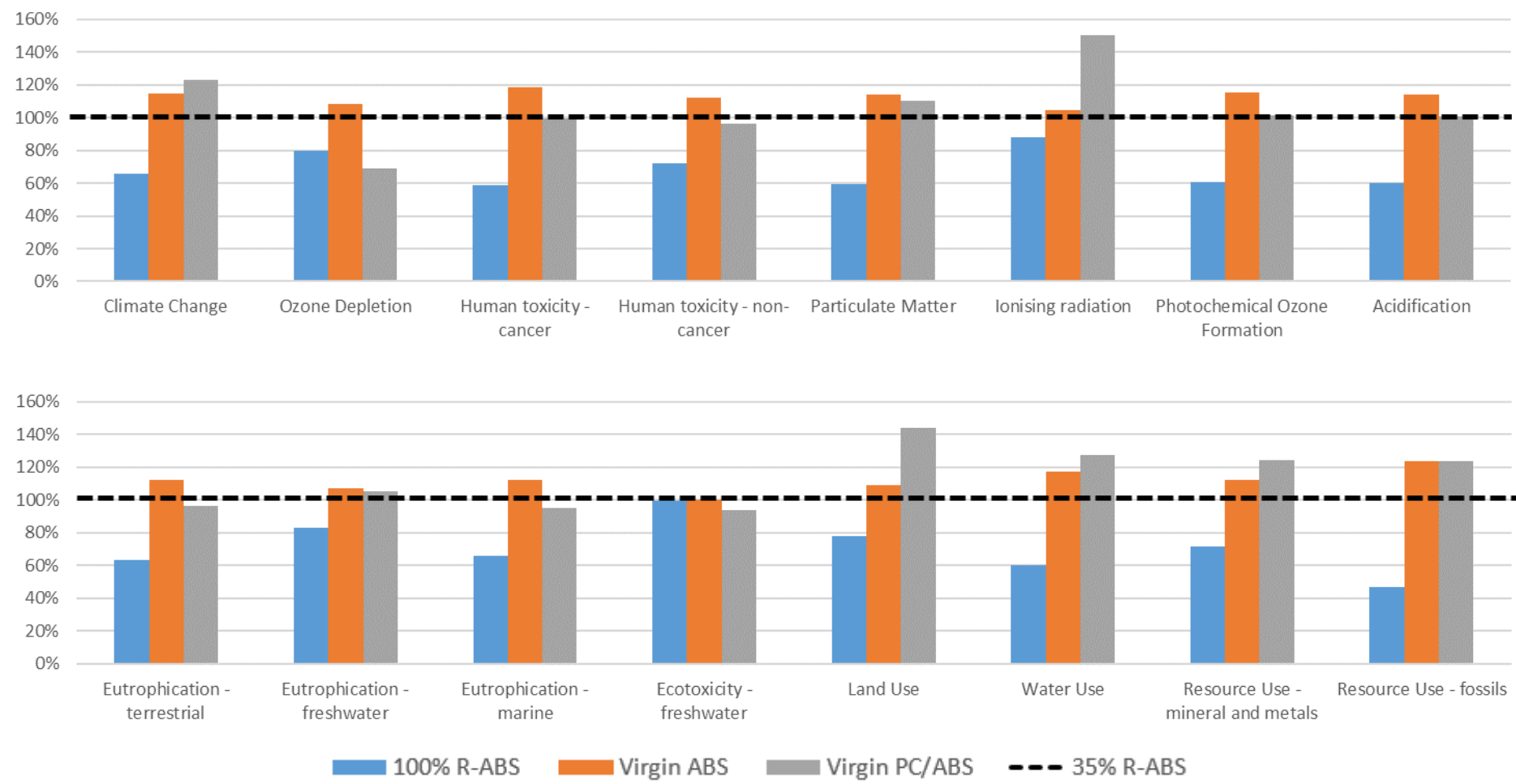
This section explores the effects of increasing the recycled content of ABS and PC/ABS printer housing panels to 100%. As a base case, this parameter was defined in the attempt to reflect the current average recycled content in electronics plastics at the EU level, and set to 35%<sup>203</sup>. However, examples of 100% recycled grades of both ABS and PC/ABS granulate are available on the market (albeit possibly obtained with the use of proper additives to achieve suitable material properties; Schwesig and Riise, 2016). A sensitivity analysis was thus performed to assess the effects of having a 100% recycled content in printer panels made of such materials.

The results of the analysis are presented in Figure 11.20 for recycled ABS panels and in Figure 11.21 for recycled PC/ABS panels. In both figures, scenario impacts are expressed as a function of the impacts of the base case of the affected printer panels scenario (i.e. 35% R-ABS and PC/ABS panels, respectively), which is set as a 100% baseline. To keep the comparison within reasonable limits, only the two reference scenarios are shown in each figure beyond the scenario affected by this sensitivity analysis (i.e. R-ABS or R-PC/ABS printer panels).

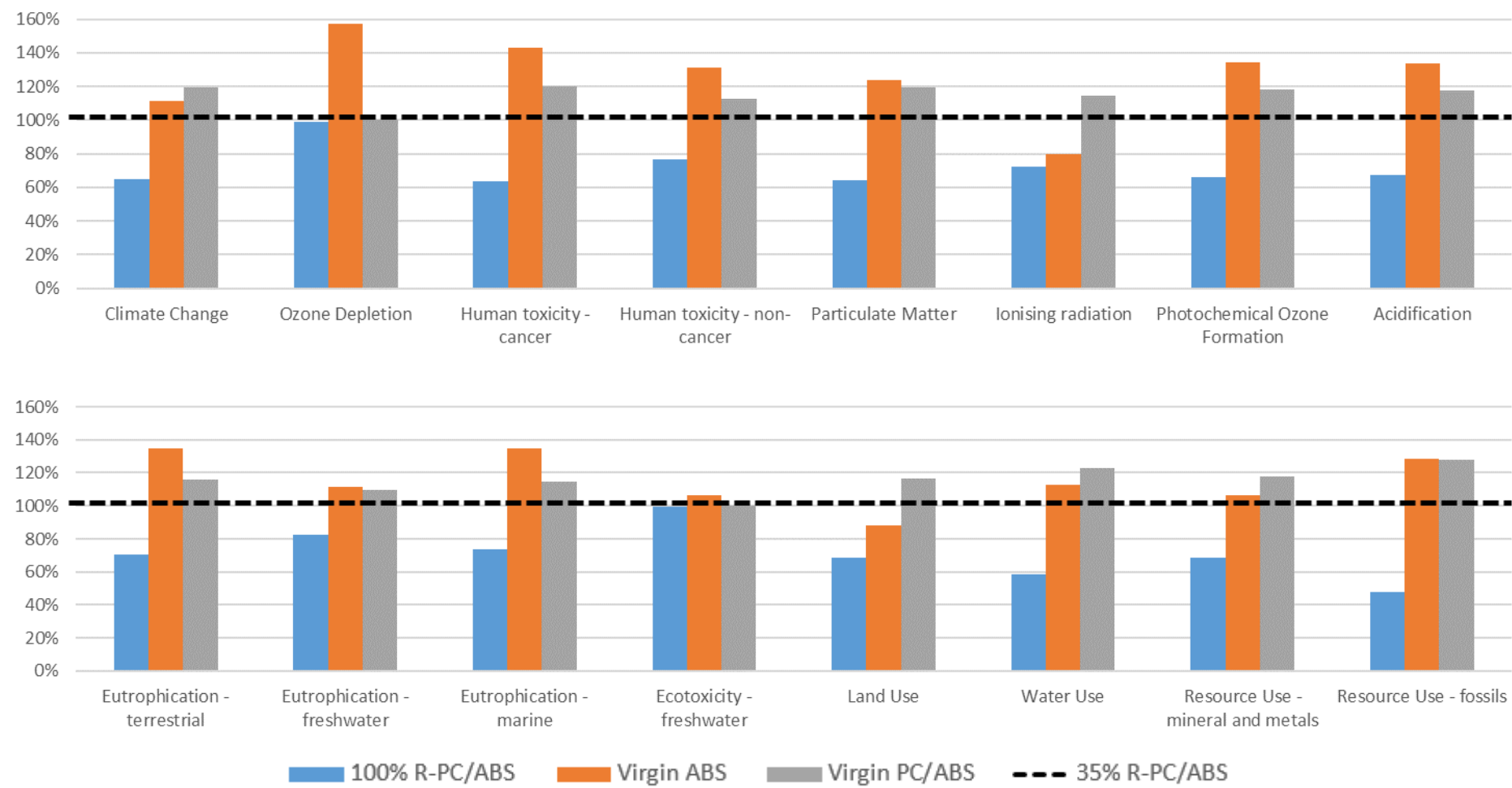
An increased (100%) content of secondary material reduces the impact of recycled ABS printer panels in all the assessed categories except for Ecotoxicity – freshwater, where no relevant changes occur compared to the base case assessment (35% recycled content). Leaving this exception apart, an impact decrease ranging from 12% (Ionising Radiation) to 53% (Resource Use – fossils) is observed. With a 100% recycled content, recycled ABS printer panels outperform their virgin counterpart across all impact categories, with an impact reduction between 16% (Ionising Radiation) and 62% (Resource Use – fossils). An exception is again represented by Ecotoxicity – freshwater, where both alternatives are still comparable. It has to be reminded, however, that the impacts of secondary ABS production are likely underestimated in this assessment (as explained in Section 11.4.2.2), so that the improvements reported above may be larger than in reality (and trade-offs may occur for certain categories).

A similar situation is observed also for 100% recycled PC/ABS printer panels, which now outperform virgin PC/ABS panels in all impact categories except for Ecotoxicity – freshwater and Ozone Depletion, where both alternatives are comparable. In this case, observed impact reductions range between 17% (Eutrophication – freshwater) and 52% (Resource Use – fossils). However, similarly to recycled ABS panels, these reductions are affected by a likely underestimate of the potential impact associated with recycled PC/ABS production.

<sup>203</sup> As discussed in Section 11.1, this value represents an estimate of the recycling rate of WEEE plastics at the EU level, and was used as an approximation for the recycled content, in the absence of more specific and representative data on the latter.



**Figure 11.20.** Results of the sensitivity analysis on the recycled material content in ABS printer housing panels, increased from 35% to 100%. Scenario impacts are expressed as a percentage of the impacts of the base case of the affected scenario (i.e. 35% R-ABS printer panels), which is set as a baseline.



**Figure 11.21.** Results of the sensitivity analysis on the recycled material content in PC/ABS printer housing panels, increased from 35% to 100%. Scenario impacts are expressed as a percentage of the impacts of the base case of the affected scenario (i.e. 35% R-PC/ABS printer panels), which is set as a baseline.

### **11.7.8.3 Feedstock origin and production country of PLA**

This sensitivity analysis evaluates the effects of relying on maize cultivated in Europe rather than in the US as a feedstock for the production of PLA used in both “neat” PLA and PLA/PC printer panels. As a consequence, also PLA production was assumed to take place in Europe and not in the US (as considered in the base case base assessment). For modelling purposes, the previously applied EF dataset for maize cultivation in the US was replaced with the respective European equivalent, while keeping the same 100 km distance for subsequent lorry transport of maize to downstream processing. The diesel input to such transport was also adjusted to reflect the average diesel supply mix for Europe, rather than for US. The dataset applied to model PLA production from maize already reflected background European conditions, so that no adjustments were performed.

The results of the analysis are presented in Figures 11.22 (for PLA printer panels) and 11.23 (for PLA/PC printer panels), setting the base case of the two printer panels scenarios (considering maize cultivation and PLA production in the US) as a 100% baseline.

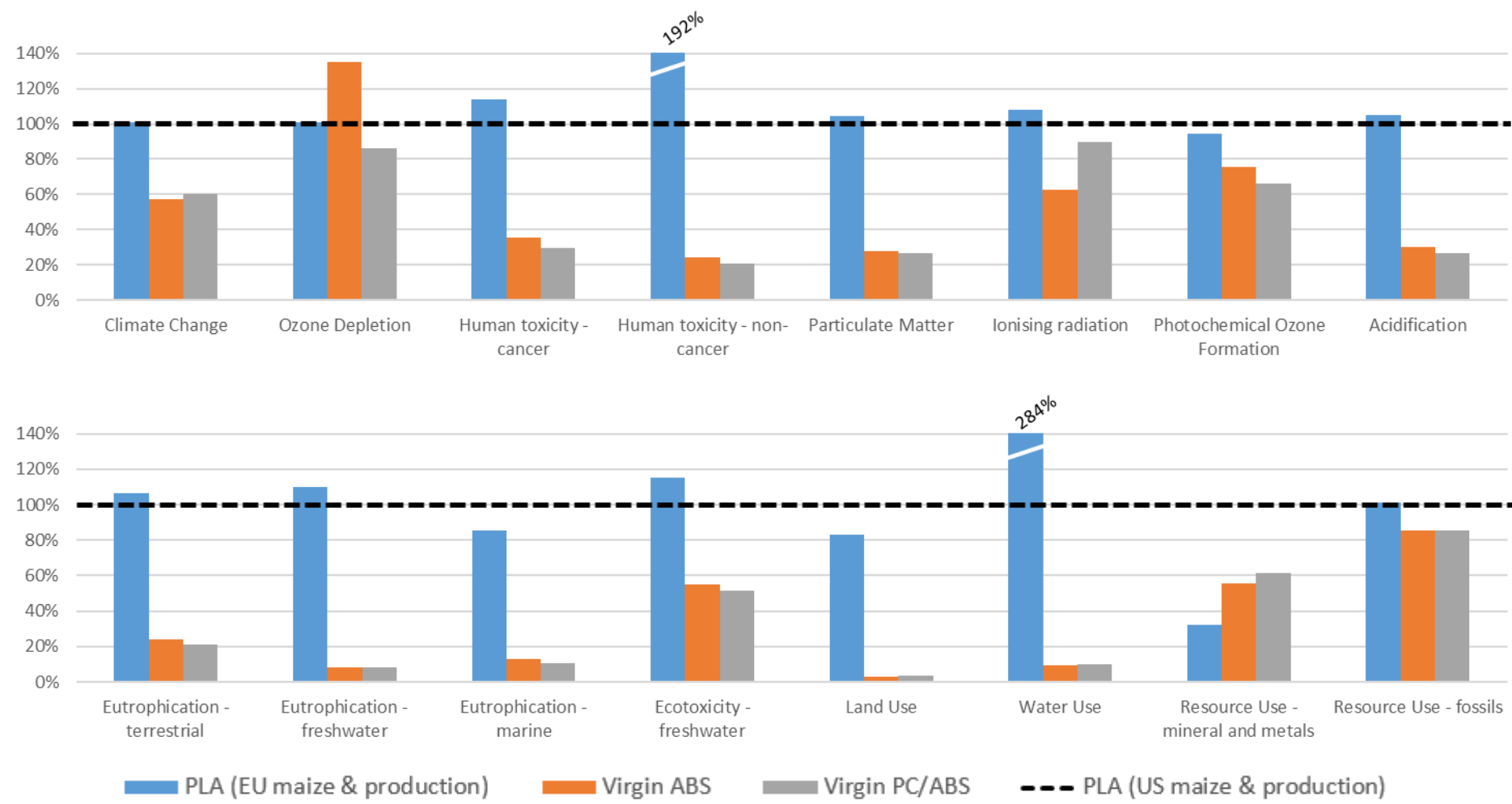
In more than half impact categories, the use of PLA produced in Europe from maize cultivated in the same region implies no relevant changes in the impact of PLA printer panels. Only in three categories the impact is reduced compared to US-based production and feedstock sourcing, mainly thanks to avoided transport from the US to Europe. These categories include Resource Use – mineral and metals (-132%), Land Use (-17%), and Eutrophication – marine (-15%). However, an impact increase is observed for toxicity-related categories (+14-92%) and Water Use (+184%). As a results, the comparison with the fossil-based alternative of reference (i.e. virgin ABS panels) is not heavily affected by the investigated change: PLA panels still show a worse performance across all impact categories except for Ozone Depletion (-19% impact reduction) and also Resource Use – minerals and metals (where the impact is now significantly decreased by 165%).

A very similar situation is observed also for PLA/PC printer panels, the only difference being that lower impact variations are observed compared to the base case assessment (assuming maize cultivation and PLA production in the US) due to the lower PLA share in the polymer blend (i.e. 65% only). This implies a reduced impact decrease in the few impact categories where the explored change of origin is beneficial<sup>204</sup> (-16-92%), and a lower increase in all the remaining categories, where such a change is detrimental (+11-73%). Also in this case, no relevant changes occur in the comparison with virgin PC/ABS panels considered as a reference, since the latter still show a better performance in all impact categories except for Ozone Depletion (where both alternatives are comparable) and also Resource Use – minerals and metals (which now shows an important impact reduction equalling 88%).

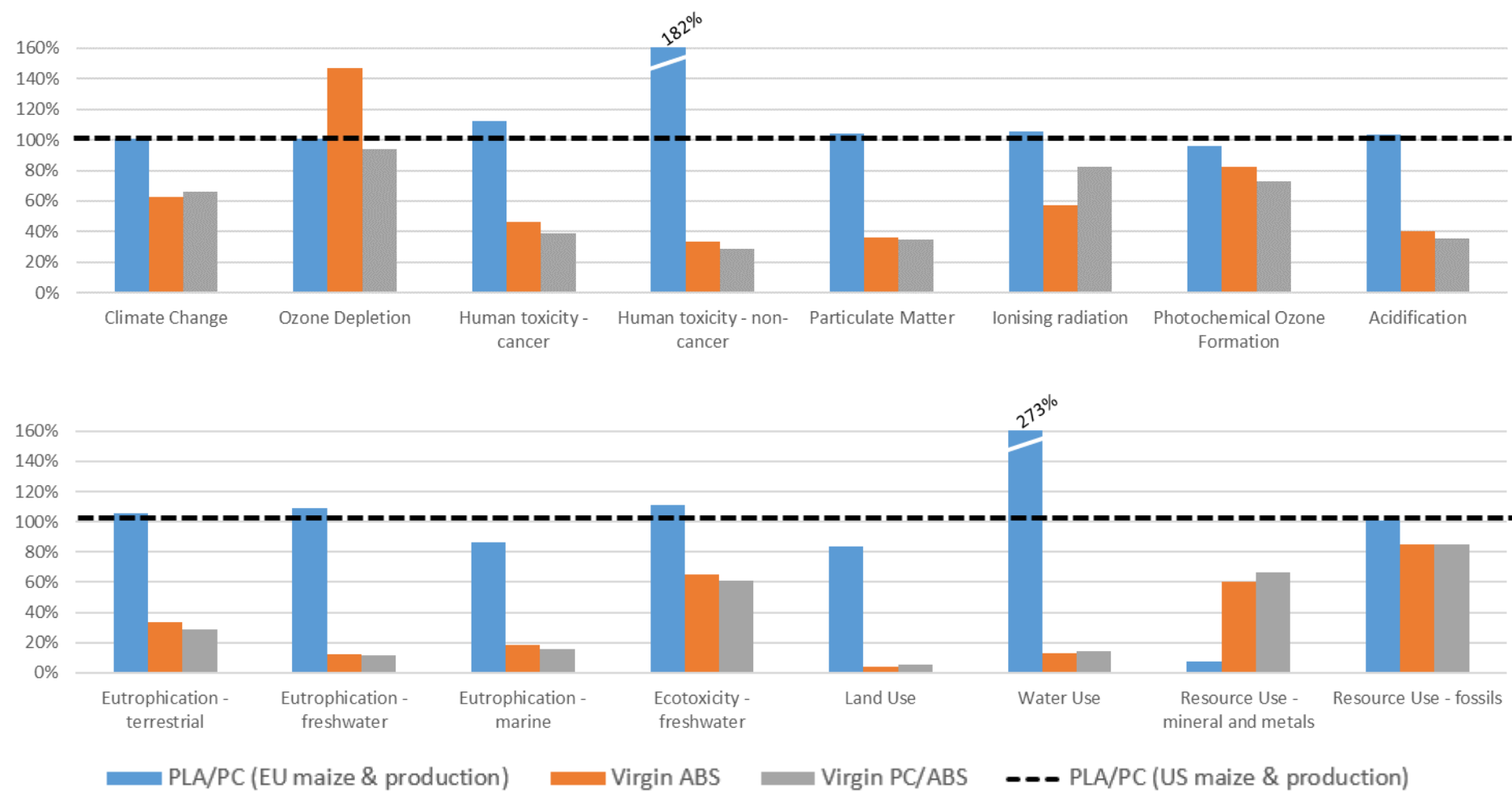
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<sup>204</sup> Resource Use – mineral and metals, Land Use and Eutrophication – marine, as observed for neat PLA printer panels.





**Figure 11.22.** Results of the sensitivity analysis on feedstock sourcing for PLA printer housing panels. Scenario impacts are expressed as a percentage of the impacts of the base case of the affected scenario (i.e. PLA printer panels from PLA produced in the US from maize grown in the same country), which is set as a baseline. Note that in some impact categories a part of the results is out of scale and is curtailed.

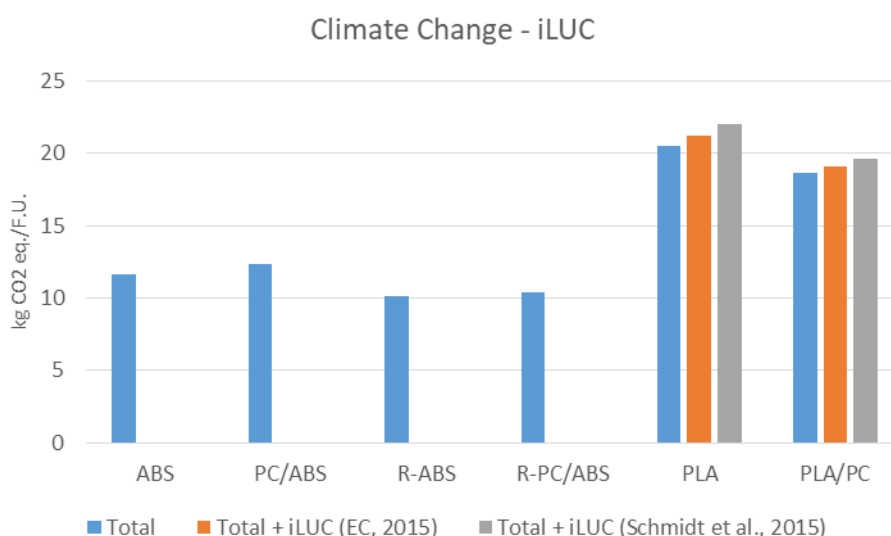


**Figure 11.23.** Results of the sensitivity analysis on feedstock sourcing for PLA/PC printer housing panels. Scenario impacts are expressed as a percentage of the base case of the affected scenario (i.e. PLA panels from PLA produced in the US from maize grown in same country), which is set as a baseline. Note that in some impact categories a part of the results is out of scale and is curtailed.

#### 11.7.8.4 iLUC model and factors

This sensitivity analysis explores the use of an alternative model (i.e. Schmidt et al., 2015) to quantify the iLUC contribution to the Climate Change impact of bio-based printer housing panels (i.e. PLA and PLA/PC panels). The iLUC factors calculated through the mentioned model were applied in place of the GHG emission factors from EC (2015) applied as a base case (Table 11.12). The results are displayed in Figure 11.24, limited to the Climate Change category, which is the only one affected by this sensitivity analysis.

Overall, the application of alternative iLUC factors only marginally affects the results. The highest impact increase observed when applying the EC (2015) factors was equal to 3.5% (maize-based PLA printer panels), while applying the model from Schmidt et al. (2015) leads to an almost doubled increase of 7.3%. However, despite this non-negligible relative increase (+111% compared to the base case), the iLUC contribution to the overall Climate Change impact of such an alternative is not significantly affected, and remains limited. For PLA/PC printer panels, the iLUC contribution calculated by applying the EC (2015) factors was lower (2.4%), because of the only partial bio-based content in the polymer blend. Applying the model from Schmidt et al. (2015), the contribution raises to 5.1%, corresponding to a relative increase comparable to that observed for PLA panels (+111%). Also in this case, the consequences of the observed increase on the overall Climate Change impact are anyway not significant. In conclusion, the results of this case study can be considered reasonably robust with respect to the estimated iLUC contribution (which is modest or negligible), and the comparison among the affected printer panels scenarios and the respective fossil-based scenarios of reference is not affected by the application of alternative factors.



**Figure 11.24.** Results of the sensitivity analysis on the applied models to quantify the iLUC contribution to the total Climate Change impact of bio-based printer housing panels.

#### 11.7.8.5 Additives in printer housing panels

This sensitivity analysis evaluates the effects of inclusion of additives in the life cycle assessment of printer housing panels scenarios. Since the product specific additive data was not available, the additives used for printer housing panels were combined from literature (OECD, 2009: data per polymer and different product groups), REACH registration tables (ECHA, 2019: data per polymer), other literature, and other suitable sources (e.g. expert opinions).

In this sensitivity analysis, the data was compiled by following steps:

1. Identification of the additive groups (e.g. flame retardants, heat stabilisers) used for specific polymers included in printer housing panels scenarios using OECD (2009) report, or other sources when data was not available in OECD (2009) (i.e. in case of recycled and bio-based polymers, see details below).
2. After identification of additive groups used for each polymer, the specific additives and their consumption amounts were searched from ECHA (2019) tables. Additives with the highest and lowest Ecotoxicity score<sup>205</sup> were selected as a representative additives.
3. Characterisation factors in EF 3.0 were used to calculate the impacts of additives using different compartments, i.e. emissions to freshwater; water, unspecified; air, unspecified; soil, unspecified, with the assumption that 100% of additive eventually ends to that compartment, even though that is not a case in reality.

In case of flame retardants, brominated flame retardant had the highest score, but because of the current tendency not to use brominated flame retardants, also the non-brominated flame retardant with highest score was included in the assessment. In case of PLA and PLA/PC blend, it was assumed that only non-brominated flame retardants are used.

In some cases ECHA (2019) tables did not contain the additives for specific polymer (e.g. there was not any heat stabilisers for ABS), although according to OECD (2009) that additive type is used. In those cases the additives in that group used for other types of polymers were used. In addition, in some cases ECHA (2019) tables did not include additive group specified in OECD (2009) or other used sources. If there was not any information available related to specific additive group (i.e. names of substances), the additive was excluded from the assessment.

Tables 11.17 and 11.18 present the additives and their use concentrations included in the assessment for ABS and PC/ABS printer panels. In case of polymer blends (i.e. PC/ABS and PLA/PC blend), the additives of both polymers were included in the assessment using the shares of polymers in the blend.

**Table 11.17.** Additives included in ABS printer housing panel scenario.

Additive group <sup>(1)</sup>	Score level	Name of substance	Concentration, % <sup>(2)</sup>
Flame retardants	Highest score	2,2',6,6'-tetrabromo-4,4'-isopropylidenediphenol	7.5% (5%-10%)
	Highest score of non-brominated	Diantimony trioxide	8%
	Lowest score	Magnesium hydroxide	6% (4%-8%)
Heat stabilisers	Highest score	2,2-dioctyl-1,3,2-oxathiaastannolan-5-one <sup>(3)</sup>	2%
	Lowest score	2-ethylhexyl 10-ethyl-4-[[2-[(2-ethylhexyl)oxy]-2-oxoethyl]thio]-4-octyl-7-oxo-8-oxa-3,5-dithia-4-stannatetradecanoate <sup>(3)</sup>	2%
Colourants	Highest score	3,6-Bis(4-tert-butylphenyl)-2,5-dihydropyrrolo[3,4-c]pyrrole-1,4-dione	2%
	Lowest score	2,9-bis[4-(phenylazo)phenyl]anthra[2,1,9-def:6,5,10-d'e'f']diisoquinoline-1,3,8,10(2H,9H)-tetrone	2%

<sup>(1)</sup> Additives used for ABS brown goods (OECD, 2009)

<sup>(2)</sup> Ranges mentioned in ECHA (2019) tables are in parenthesis, averages used in assessment

<sup>(3)</sup> Not specified to be used for ABS in ECHA (2019) tables

<sup>205</sup> Score was calculated multiplying use concentration (maximum if range) with Ecotoxicity – freshwater characterisation factors in EF 3.0 method (emissions to freshwater).

1 **Table 11.18.** Additives included in PC/ABS blend printer housing panel scenario.

Additive group <sup>(1)</sup>	Score level	Name of substance	Concentration, % <sup>(2)</sup>
<b>ABS part in blend</b>			
Flame retardants	Highest score	2,2',6,6'-tetrabromo-4,4'-isopropylidenediphenol	7.5% (5%-10%)
	Highest score of non-brominated	Diantimony trioxide	8%
	Lowest score	Magnesium hydroxide	6% (4%-8%)
<b>PC part in blend</b>			
Coupling agents <sup>(3)</sup>	-	-	-
Blowing agents <sup>(4)</sup>	-	Azodicarbonamide <sup>(5)</sup>	0.1%
Lubricants	Highest score	Glycerides, C16-18 mono- and di-	1%
	Lowest score	Zinc distearate	0.75%
Fillers <sup>(4)</sup>	-	Cobalt bis(2-ethylhexanoate) <sup>(5)</sup>	17.5% (15%-20%)
<b>Whole PC/ABS blend</b>			
Heat stabilisers	Highest score	2,2-dioctyl-1,3,2-oxathiastannolan-5-one <sup>(5)</sup>	2%
	Lowest score	2-ethylhexyl 10-ethyl-4-[[2-[(2-ethylhexyl)oxy]-2-oxoethyl]thio]-4-octyl-7-oxo-8-oxa-3,5-dithia-4-stannatetradecanoate <sup>(5)</sup>	2%
Colourants	Highest score	3,6-Bis(4-tert-butylphenyl)-2,5-dihydropyrrolo[3,4-c]pyrrole-1,4-dione	2%
	Lowest score	2,9-bis[4-(phenylazo)phenyl]anthra[2,1,9-def:6,5,10-d'e'f']diisoquinoline-1,3,8,10(2H,9H)-tetrone	2%

(1) Additives used for ABS brown goods or PC electric devices (OECD, 2009)

(2) Ranges mentioned in ECHA (2019) tables are in parenthesis, averages used in assessment

(3) Not any coupling agents in ECHA (2019) tables, left out from the assessment

(4) Only one substance with CF available

(5) Not specified to be used for ABS or PC in ECHA (2019) tables

There was not any data available for recycled polymers in OECD (2009) report, thus additional literature data was used to assess what additives are needed for them. According to Peydro et al. (2013), styrene-ethylene/butylene-styrene (SEBS) can be used for recycled ABS to achieve similar mechanical properties with virgin polymer. According to their study, 5% concentration would already be enough. However, Van Damme et al. (2016) studied styrene-ethylene/butylene-styrene grafted with maleic anhydride (SEBS-g-MA), and they concluded that starting from 20%, the big increase in quality can be seen. In this study, it was assumed that recycled polymers contain same additives as same virgin polymers (e.g. recycled ABS contains same additives with virgin ABS), and in addition 5%, 10% or 20% of SEBS-g-MA for both recycled ABS and recycled PC/ABS blend. Characterisation factor for SEBS-g-MA was combined using different components of additive, i.e. styrene, ethylene, butylene and maleic anhydride. However, CF for butylene is not included in EF 3.0, so butylene glycol was used instead.

In addition, there was not any data related to bio-based polymers in OECD (2009), thus other literature sources and expert opinions had to be used. According to Molenveld (2019), nucleating agents (commonly Talcum) and impact modifiers are used for crystalline PLA. Impact modifiers for PLA are offered by Dow and Arkema (e.g. Paraloid or Biostrength), but there was not any detailed information of the substances in those additives. Also, ECHA (2019) did not include any impact modifiers, and thus impact modifiers were excluded from the assessment. In case of nucleating agents, also those included in the ECHA (2019) tables were included in the assessment for comparison, even it is not sure if they can be used for PLA. In addition, flame retardants were included in the assessment (Table 11.19). In PLA/PC blend, the same additives as for crystalline PLA were assumed for the PLA part of the blend, and additives used for PC where assumed for PC part of the blend (Table 11.20).

1 **Table 11.19.** Additives included in PLA printer housing panel scenario.

Additive group <sup>(1)</sup>	Score level	Name of substance	Concentration, % <sup>(2)</sup>
Flame retardants	Highest score (non-brominated)	Diantimony trioxide	8%
	Lowest score	Magnesium hydroxide	6% (4%-8%)
Nucleating agents	Recommended	Talcum	5.2% (3.7%-6.7%) <sup>(3)</sup>
	Highest score	2,2'-methylene bis-(4,6-di-tert.butylphenyl) sodium phosphate <sup>(4)</sup>	0.2%
	Lowest score	Sodium benzoate <sup>(4)</sup>	0.2%
Impact modifiers <sup>(5)</sup>	-	-	-

2 <sup>(1)</sup> Additives used for PLA (Molenveld, 2019)

3 <sup>(2)</sup> Ranges mentioned in ECHA (2018) tables are in parenthesis, averages used in assessment

4 <sup>(3)</sup> Concentration based on Broeren et al. (2016) where share of additives in printer housing panels were 11-20%, dividing that for three additive types used, the average concentration for one would be 5.2%

5 <sup>(4)</sup> Not specified to be used for PLA in ECHA (2019) tables

6 <sup>(5)</sup> Not any information available substances used as impact modifiers for PLA, and not any information available in ECHA (2019) tables, left out from the assessment

1 **Table 11.20.** Additives included in PLA/PC printer housing panel scenario.

Additive group <sup>(1)</sup>	Score level	Name of substance	Concentration, % <sup>(2)</sup>
<b>PLA part in the blend</b>			
Flame retardants	Highest score (non-brominated)	Diantimony trioxide	8%
	Lowest score	Magnesium hydroxide	6% (4%-8%)
Nucleating agents	Recommended	Talcum	5.2% (3.7%-6.7%) <sup>(3)</sup>
	Highest score	2,2'-methylene bis-(4,6-di-tert.butylphenyl) sodium phosphate <sup>(4)</sup>	0.2%
	Lowest score	Sodium benzoate <sup>(4)</sup>	0.2%
Impact modifiers <sup>(5)</sup>	-	-	-
<b>PC part in the blend</b>			
Coupling agents <sup>(6)</sup>	-	-	-
Blowing agents <sup>(7)</sup>	-	Azodicarbonamide <sup>(8)</sup>	0.1%
Lubricants	Highest score	Glycerides, C16-18 mono- and di-	1%
	Lowest score	Zinc distearate	0.75%
Fillers <sup>(7)</sup>	-	Cobalt bis(2-ethylhexanoate) <sup>(8)</sup>	17.5% (15%-20%)
Heat stabilisers	Highest score	2,2-di-octyl-1,3,2-oxathia-stannolan-5-one <sup>(8)</sup>	2%
	Lowest score	2-ethylhexyl 10-ethyl-4-[[2-[(2-ethylhexyl)oxy]-2-oxoethyl]thio]-4-octyl-7-oxo-8-oxa-3,5-dithia-4-stannatetradecanoate <sup>(8)</sup>	2%
Colourants	Highest score	3,6-Bis(4-tert-butylphenyl)-2,5-dihydropyrrolo[3,4-c]pyrrole-1,4-dione	2%
	Lowest score	2,9-bis[4-(phenylazo)phenyl]anthra[2,1,9-def:6,5,10-d'e'f']diisoquinoline-1,3,8,10(2H,9H)-tetrone	2%

2 <sup>(1)</sup> Additives used for PLA (Molenveld, 2019)

3 <sup>(2)</sup> Ranges mentioned in ECHA (2018) tables are in parenthesis, averages used in assessment

4 <sup>(3)</sup> Concentration based on Broeren et al. (2016) where share of additives in printer housing panels were 11-20%, dividing that for three additive types used, the average concentration for one would be 5.2%

5 <sup>(4)</sup> Not specified to be used for PLA in ECHA (2019) tables

6 <sup>(5)</sup> Not any information available substances used as impact modifiers for PLA, and not any information available in ECHA (2019) tables, left out from the assessment

7 <sup>(6)</sup> Not any coupling agents in ECHA (2019) tables, left out from the assessment

8 <sup>(7)</sup> Only one substance with CF available

9 <sup>(8)</sup> Not specified to be used for PC in ECHA (2019) tables

12 Figures 11.25-11.36 presents the results of the printer housing panels scenarios (i.e. ABS, PC/ABS, R-ABS, R-PC/ABS, PLA and PLA/PC) for Ecotoxicity – freshwater and Human Toxicity – non-cancer impact categories. In addition Figures 11.37 and 11.38 presents the comparison between different polymers. Note that in the figures base case results are characterised with EF 2.0 method (as all base case results in this report), but in case of additives, EF 3.0 characterisation factors are used, because the lack of characterisation factors for chemicals in EF 2.0. Note also, that the impact of additives includes only impacts after use, not the impacts of production or leakage in the polymer production site.

21 In all included scenarios, Ecotoxicity – freshwater impact is highest when all additives were assumed to end up to freshwater compartment and lowest with soil, unspecified compartment. In case of maximum scenarios (i.e. using additives with highest impact score) the impact of additives are multiple compared to base case impacts, i.e. base case impacts are usually less than 1% when using characterisation factors for freshwater or

water, unspecified compartments. Only exception is PLA, which base case impacts are 70-90%. However, it should be noted that in case of PLA, 1/3 of the additives are excluded because of the lack of data. When using characterisation factors for air and soil (unspecified) compartments, the impact of base case is more contributing. In case of PLA almost 100% of the impacts are due to base case, and for other polymers up to 50%.

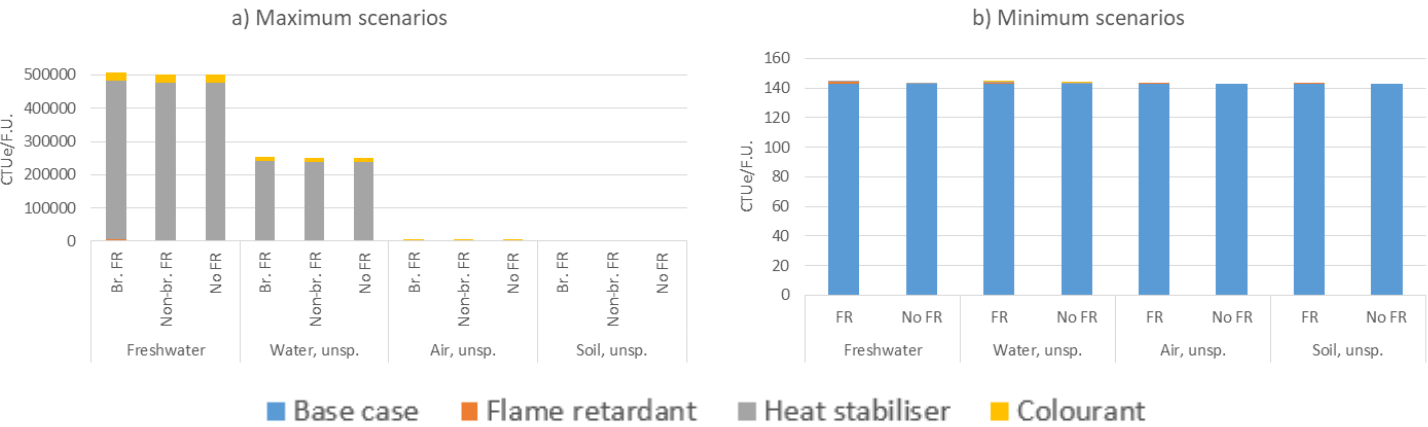
In contrary, in minimum scenarios of Ecotoxicity – freshwater, the impact of additives are not usually significant compared to base case impacts. However, SEBS-g-MA used for recycled polymers (R-ABS and R-PC/ABS) has significant impact when using the highest concentration of SEBS-g-MA (i.e. 20%), and characterisation factors for freshwater or water, unspecified compartment (84% and 75% shares of SEBS-g-MA from total impact). Also with the lowest SEBS-g-MA concentration (5%) the impact would be significant with CF's for freshwater and water, unspecified compartment, 21% and 18% of total Ecotoxicity – freshwater impact. In addition, printer panels containing PC (PC/ABS, R-PC/ABS, PLA/PC) have significant impact due to fillers used in PC part. This is at least partially due to the fact that there was only one filler available in the ECHA (2019) database, thus same filler is used in both minimum and maximum scenarios. It should also be noted that according to ECHA (2019) database, that filler is not used for PC, or more specifically PC was not included in the polymers that filler is typically used.

In Human Toxicity – non-cancer impact category, the similar picture can be noticed, i.e. in maximum scenarios the impact of additives are dominating, while in minimum scenarios impact of base cases are dominating, with some additives showing significant contribution, e.g. fillers in polymers containing PC, heat stabilisers when using CF's for air, unspecified compartment, and nucleating agent (other than talcum) for PLA. For Human Toxicity – non-cancer, the impacts are usually highest when characterisation factors for air, unspecified compartment are used.

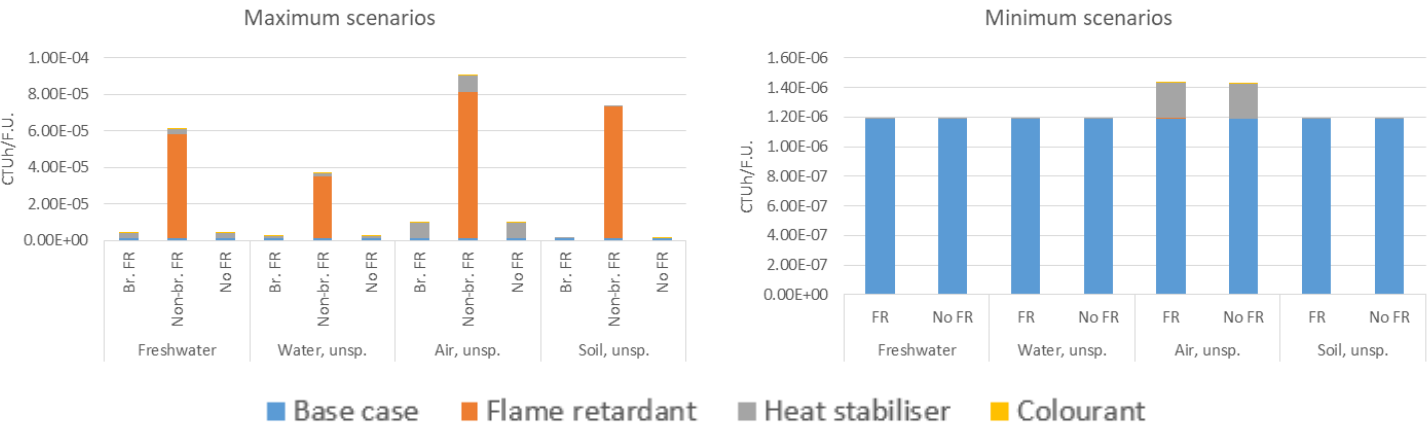
The most contributing additive varies between polymers, scenarios and impact category. In case of fossil and recycled printer panels (ABS, PC/ABS, R-ABS, R-PC/ABS), as well as maximum scenarios of PLA/PC, heat stabiliser have the highest impact in Ecotoxicity – freshwater, while non-brominated flame retardant has the highest impact in maximum scenarios of Human Toxicity – non-cancer. In Ecotoxicity – freshwater impact, there is only slight impact if flame retardant is included or not, or if brominated flame retardant is replaced with non-brominated.

When comparing printer panels made of different polymers, Ecotoxicity – freshwater impact in the maximum scenarios (with highest impact score additives) are highest with the fossil polymers, no matter if it is virgin or recycled, as the additives assumed are almost same to both. Recycled polymers have one additive more compared to virgin ones, but the impact of that additive is negligible in maximum scenarios. PLA printer panels have a lowest impact in maximum scenarios when impact of additives is included. When looking for minimum scenarios of printer panel Ecotoxicity – freshwater impact, it can be noted that printer panels containing PC have the highest impact. This is, again, due to fillers, as explained earlier. Human Toxicity – non-cancer impacts are similar to Ecotoxicity impacts in maximum scenarios, i.e. PLA printer panels have a lowest impact. In contrary to Ecotoxicity impacts, in the minimum scenarios, PLA has the highest Human Toxicity – non-cancer impact due to highest base case impact and low share of additive impacts for all included polymers.

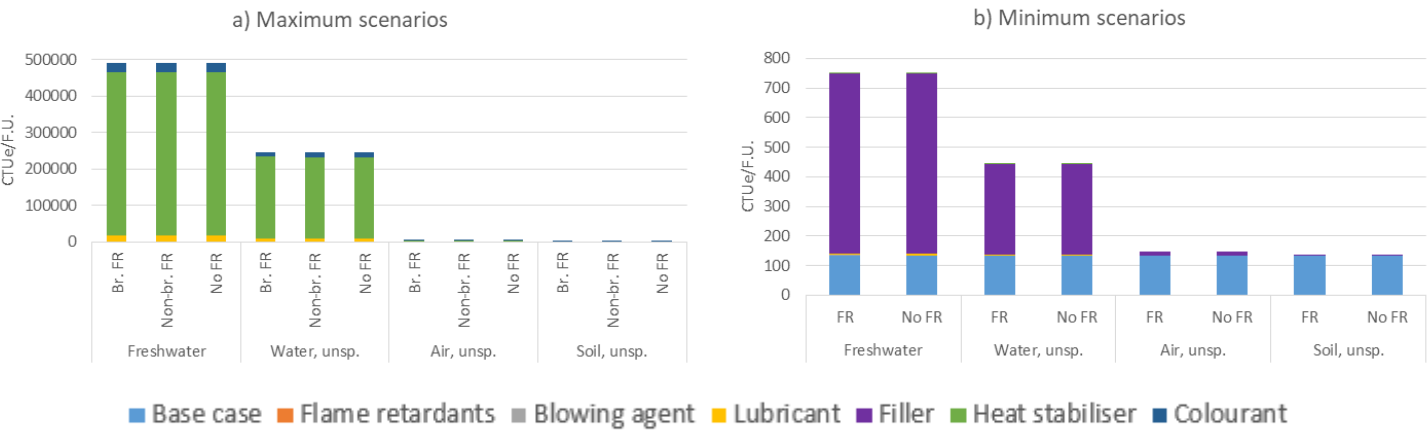




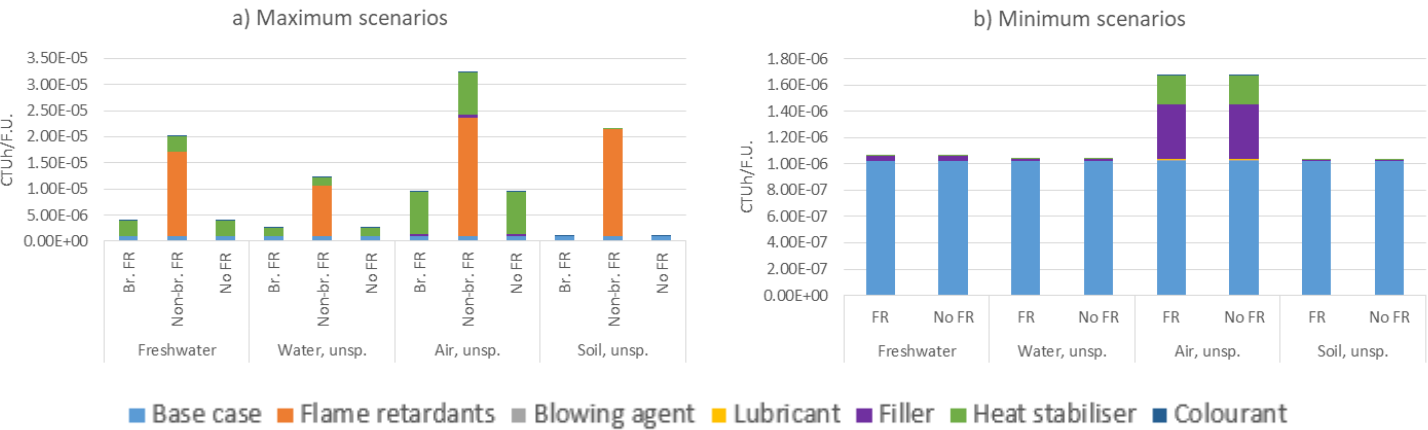
**Figure 11.25.** Ecotoxicity – freshwater impacts of ABS printer housing panels in a) Maximum score scenarios, and b) Minimum score scenarios. Note that highest impact score for flame retardants was for brominated flame retardant (Br. FR), and thus also non-brominated flame retardant was included (Non-br. FR).



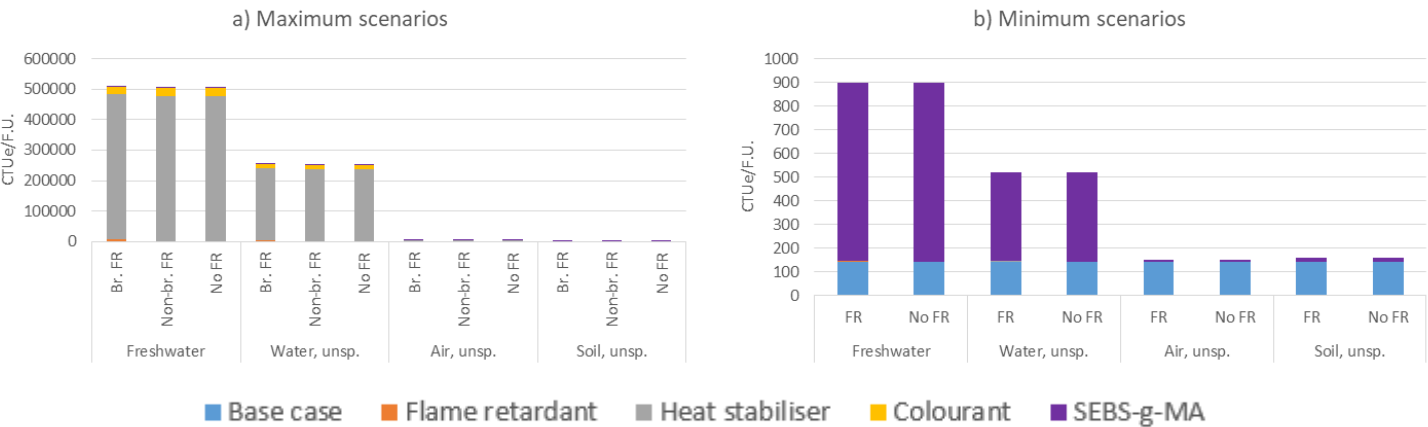
**Figure 11.26.** Human Toxicity – non-cancer impacts of ABS printer housing panels in a) Maximum score scenarios, and b) Minimum score scenarios. Note that highest impact score for flame retardants was for brominated flame retardant (Br. FR), and thus also non-brominated flame retardant was included (Non-br. FR).



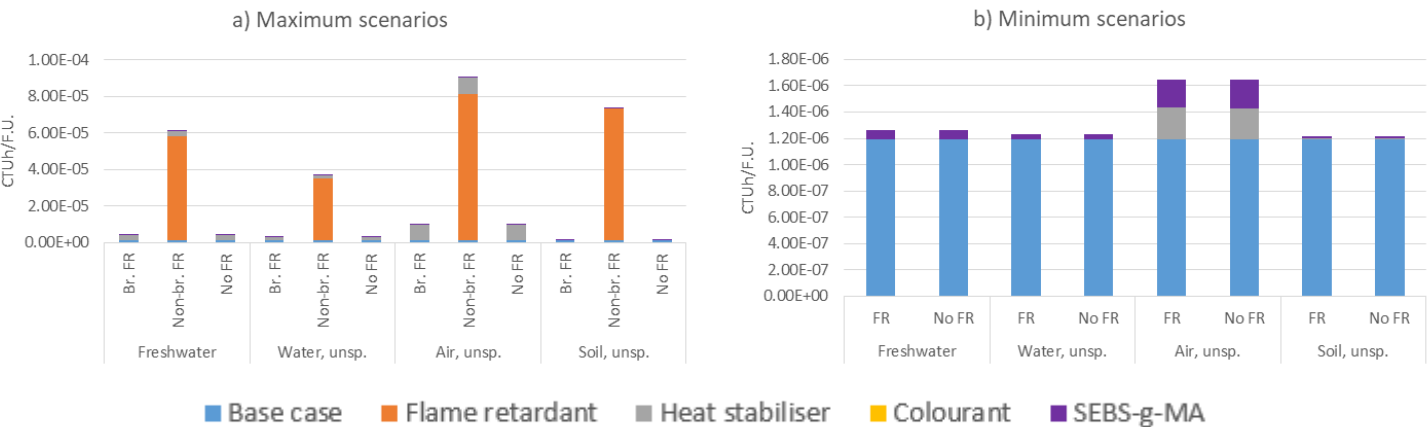
**Figure 11.27.** Ecotoxicity – freshwater impacts of PC/ABS printer housing panels in a) Maximum score scenarios, and b) Minimum score scenarios. Note that highest impact score for flame retardants was for brominated flame retardant (Br. FR), and thus also non-brominated flame retardant was included (Non-br. FR).



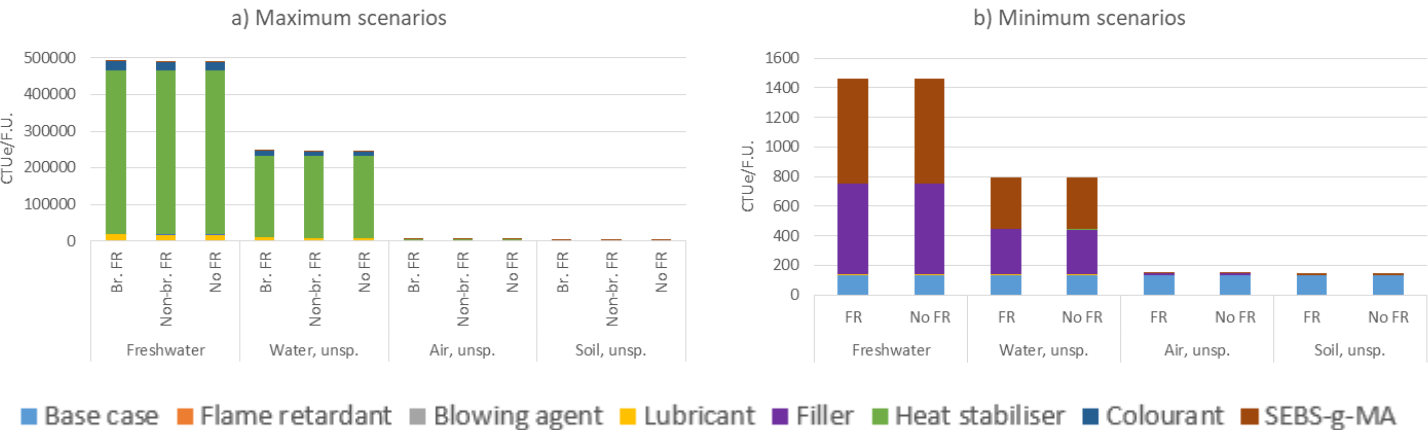
**Figure 11.28.** Human Toxicity – non-cancer impacts of PC/ABS printer housing panels in a) Maximum score scenarios, and b) Minimum score scenarios. Note that highest impact score for flame retardants was for brominated flame retardant (Br. FR), and thus also non-brominated flame retardant was included (Non-br. FR).



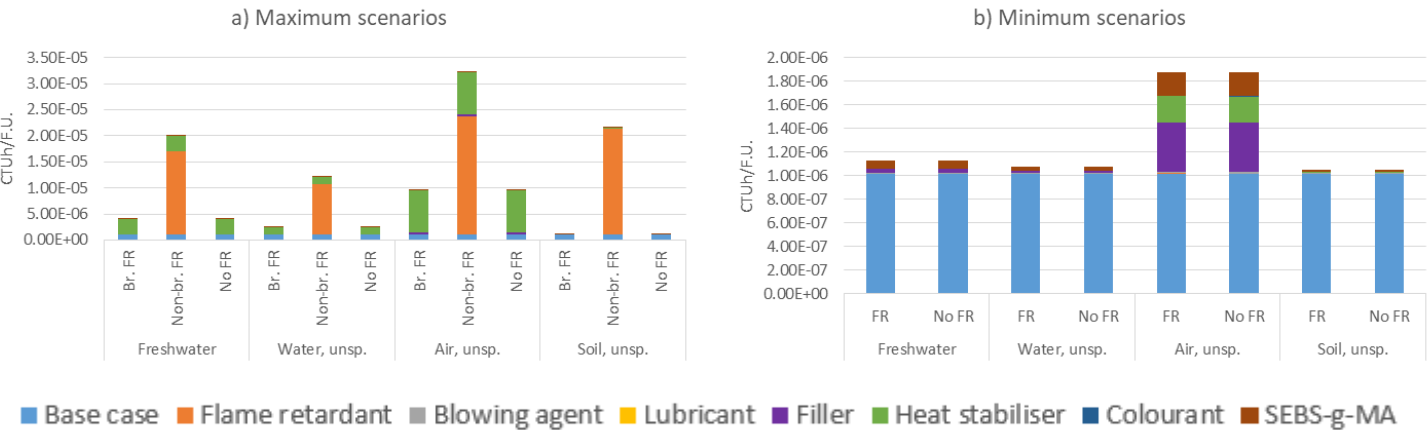
**Figure 11.29.** Ecotoxicity – freshwater impacts of R-ABS printer housing panels in a) Maximum score scenarios, and b) Minimum score scenarios. Note that highest impact score for flame retardants was for brominated flame retardant (Br. FR), and thus also non-brominated flame retardant was included (Non-br. FR).



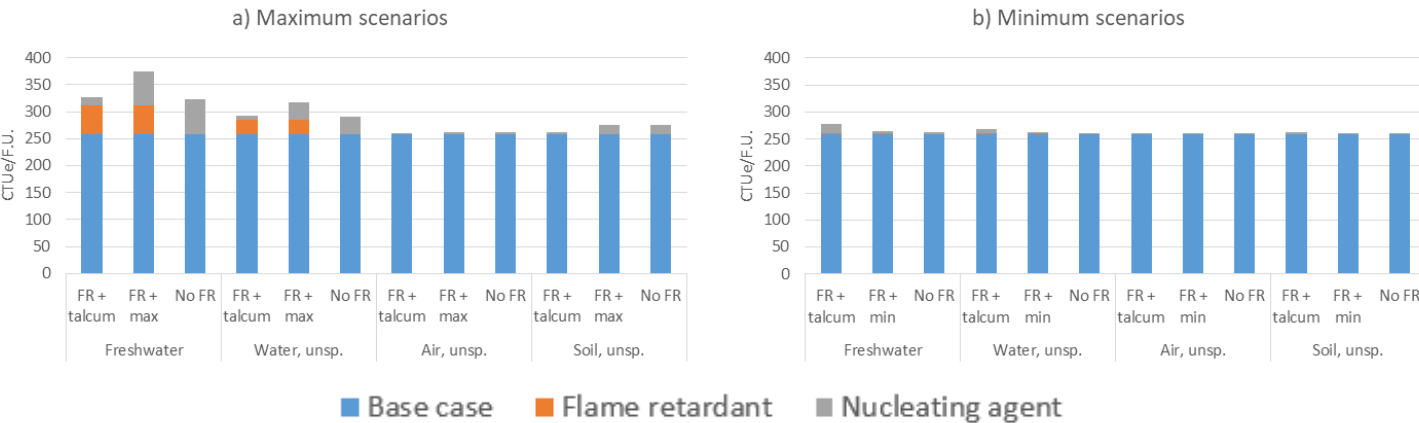
**Figure 11.30.** Human Toxicity – non-cancer impacts of R-ABS printer housing panels in a) Maximum score scenarios, and b) Minimum score scenarios. Note that highest impact score for flame retardants was for brominated flame retardant (Br. FR), and thus also non-brominated flame retardant was included (Non-br. FR).



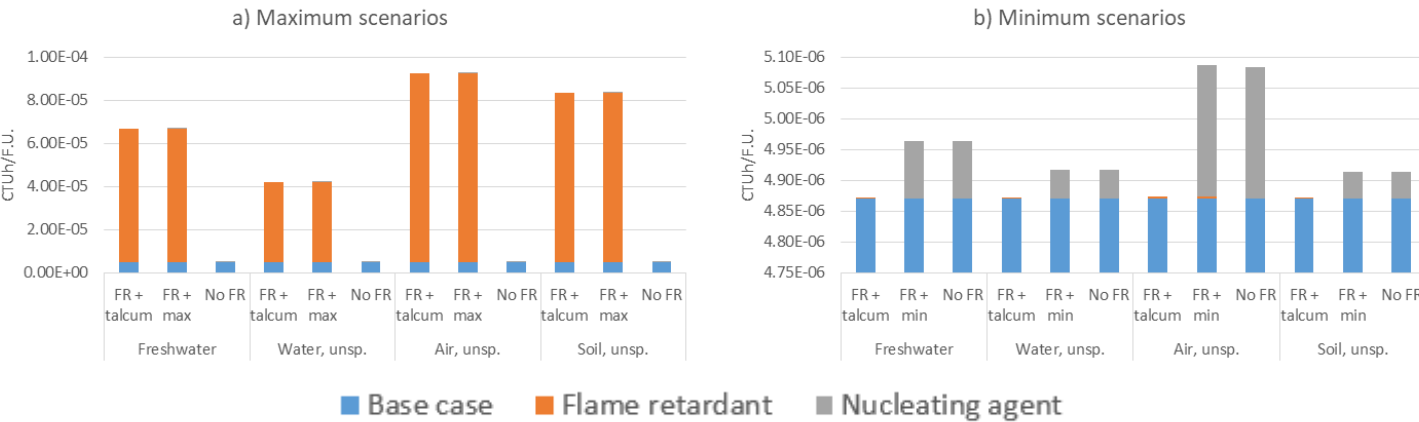
**Figure 11.31.** Ecotoxicity – freshwater impacts of R-PC/ABS printer housing panels in a) Maximum score scenarios, and b) Minimum score scenarios. Note that highest impact score for flame retardants was for brominated flame retardant (Br. FR), and thus also non-brominated flame retardant was included (Non-br. FR).



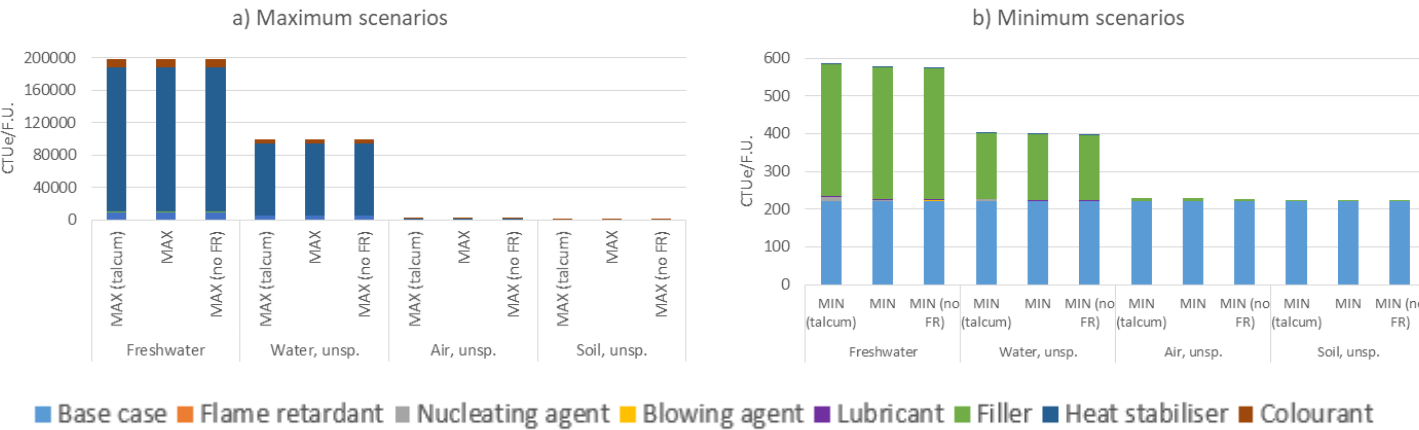
**Figure 11.32.** Human Toxicity – non-cancer impacts of R-PC/ABS printer housing panels in a) Maximum score scenarios, and b) Minimum score scenarios. Note that highest impact score for flame retardants was for brominated flame retardant (Br. FR), and thus also non-brominated flame retardant was included (Non-br. FR).



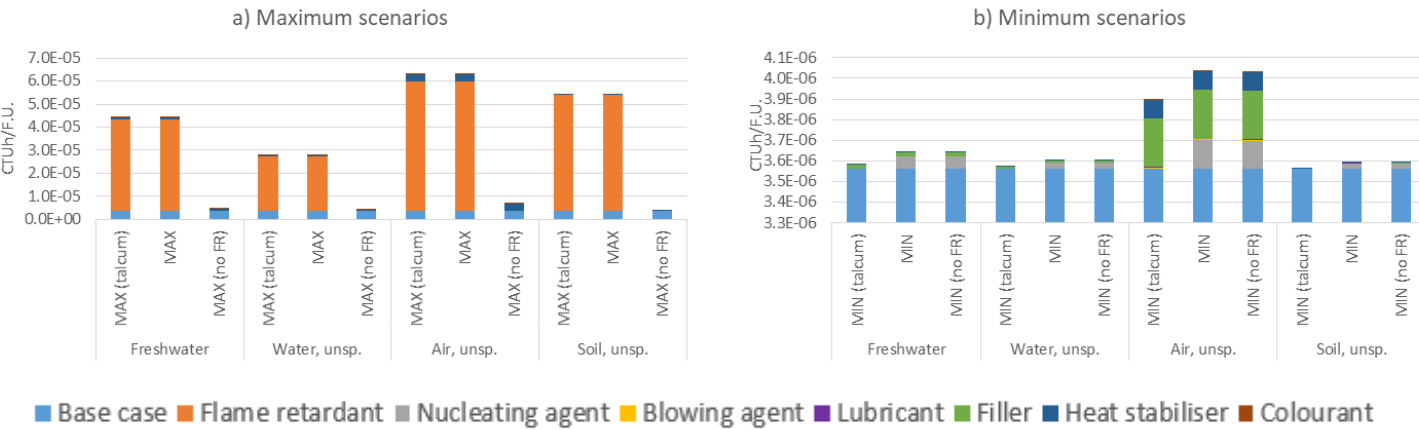
**Figure 11.33.** Ecotoxicity – freshwater impacts of PLA printer housing panels in a) Maximum score scenarios, and b) Minimum score scenarios. Note that for PLA only non-brominated flame retardant was included (FR). Nucleating agent is either talcum or conventional nucleating agent (in FR + max, FR + min, and no FR scenarios).



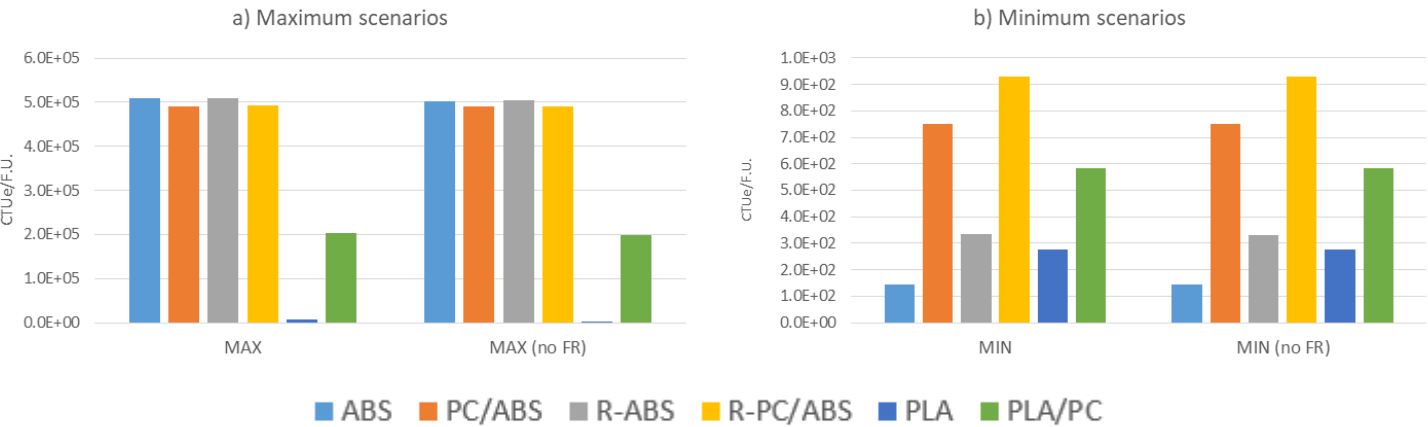
**Figure 11.34.** Human Toxicity – non-cancer impacts of PLA printer housing panels in a) Maximum score scenarios, and b) Minimum score scenarios. Note that for PLA only non-brominated flame retardant was included (FR). Nucleating agent is either talcum or conventional nucleating agent (in FR + max, FR + min, and no FR scenarios).



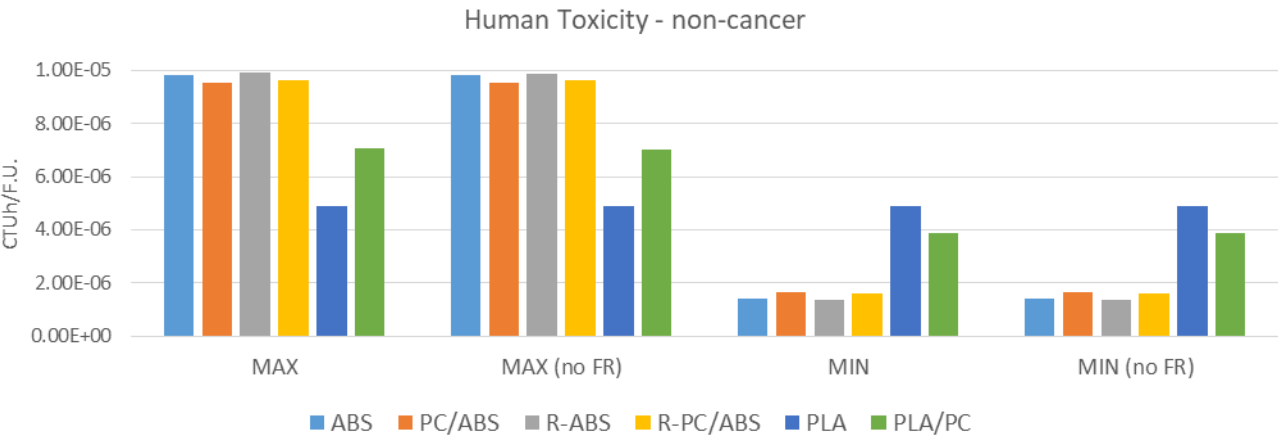
**Figure 11.35.** Ecotoxicity – freshwater impacts of PLA/PC printer housing panels in a) Maximum score scenarios, and b) Minimum score scenarios. Note that for PLA only non-brominated flame retardant was included (FR). Nucleating agent is either talcum or conventional nucleating agent (in FR + max, FR + min, and no FR scenarios).



**Figure 11.36.** Human Toxicity – non-cancer impacts of PLA/PC printer housing panels in a) Maximum score scenarios, and b) Minimum score scenarios. Note that for PLA only non-brominated flame retardant was included (FR). Nucleating agent is either talcum or conventional nucleating agent (in FR + max, FR + min, and no FR scenarios).



**Figure 11.37.** Ecotoxicity - freshwater impacts of printer housing panels LCA scenarios with a) Maximum and b) Minimum additive scenarios. Note that for PLA the scenario with talcum is selected to comparison.



**Figure 11.38.** Human Toxicity – non-cancer impacts of printer housing panels LCA scenarios. Note that for PLA the scenario with talcum is selected to comparison.

## 12 Case study 9: Monobloc stacking chairs

This case study focuses on monobloc stacking chairs for private or public use (e.g. events), as a representative article of the use of plastic materials in the furniture sector. While a variety of alternatives is available on the market, possibly consisting of other materials than plastics for the different parts (e.g. metals for reinforcing purposes, foams and textiles for cushioning etc.), for the purpose of simplicity this study focuses on simpler chairs exclusively produced by one material and without any cushioning.

### 12.1 Assessed scenarios

The use of different materials and/or feedstock sources for plastic chair manufacturing was explored by assessing a number of alternative scenarios (Table 12.1). Two reference scenarios relying on the two most commonly applied fossil-based polymers for this type of application (i.e. HDPE and PP) were analysed first. The use of recycled (fossil-based) post-consumer plastic waste as a feedstock for the production of the same polymers (R-HDPE and R-PP) was then explored (Scenarios 3 and 4, respectively), assuming a 100% recycled content. Although different shares of recycled material can be mixed with virgin material to be used as input to chairs production, this study focuses on chairs relying entirely on recycled input as an extreme case. This assumption was made in the absence of more specific and representative data on the average recycled content in this type of article (and plastic furniture in general) at the EU level, and allows assessing the effects of a complete substitution of virgin with recycled material. Two fully bio-based, drop-in alternatives to the same polymers (i.e. bio-HDPE and bio-PP) were finally assessed in Scenarios 5 and 6, respectively. In this case, bio-Ethylene or bio-Propylene derived from Brazilian sugarcane were considered as a feedstock, which is the most widely used at present for these bio-based intermediates.

**Table 12.4.** LCA scenarios assessed for the monobloc stacking chair case study.

Scenario	Polymer	Monomer(s)	Feedstock	End of Life options <sup>(1)</sup>
1 - Conventional polymer 1	HDPE	Ethylene	Fossil-based (oil/natural gas)	Recycling Incineration Landfilling
2 - Conventional polymer 2	PP	Propylene	Fossil-based (oil/natural gas)	Recycling Incineration Landfilling
3 - Alternative polymer 1	R-HDPE	Ethylene	Waste HDPE (post-consumer)	Recycling Incineration Landfilling
4 - Alternative polymer 2	R-PP	Propylene	Waste PP (post-consumer)	Recycling Incineration Landfilling
5 - Alternative polymer 3	Bio-HDPE	Bio-ethylene	Sugarcane (Brazil)	Recycling Incineration Landfilling
5 - Alternative polymer 4	Bio-PP	Bio-propylene	Sugarcane (Brazil)	Recycling Incineration Landfilling

<sup>(1)</sup> The impacts of scenarios were individually assessed for each listed End of Life option, as well as for a combination of such options reflecting as far as possible the average situation at the EU level.

### 12.2 Functional Unit and reference flow

The main function of the studied article is to provide seating support to people in private or public spaces or events. The functional unit of this case study was thus defined as “providing free seating support to one person, by means of one chair without armrests



and cushioning, which does not break or excessively discolour over a minimum duration of 10 years in the EU" (Table 12.2).

**Table 12.5.** Definition of the functional unit for monobloc stacking chairs LCA scenarios.

Aspect	Description
"What" (function(s) or service(s) provided)	Providing free seating support without armrests and cushioning
"How much" (extent of the function(s) or service(s))	For a single person
"How well" (expected level of quality)	By means of one chair with a total height of 80 cm, a seat height of 43 cm, which does not break and excessively discolour during the intended use
"How long" (duration/lifetime of the function or service)	At least 10 years
"Where" (location/geography of the service)	In the EU

The reference flow of each scenario (i.e. the amount of material required to fulfil the functional unit), was defined according to the technical details of a specific product available on the market at the time of developing this study, as suggested by one stakeholder. The latter is a stacking polypropylene chair with a total height of 80 cm (seat height 43 cm) a width of 45 cm, and a depth of 50.5 cm, corresponding to an overall mass equal to 2.29 kg. The same mass has been assumed also for Bio-PP chairs (changing feedstock does not affect material properties) and R-PP chairs, assuming recycled material has comparable properties to the replaced virgin material (possibly achieved through the use of suitable additives). Conversely, the mass of HDPE chairs (from all types of feedstock) was calculated based on the respective average material density, and assuming the same volume of polymer as PP chairs is used (i.e. both types of chair have the same dimensions). Since the two materials have comparable densities, the final mass estimated for HDPE chairs (2.40 kg) is comparable with that assumed for PP chairs. Table 12.3 summarises the reference flow in each scenario.

**Table 12.6.** Calculation of the reference flow for Monobloc stacking chairs LCA scenarios.

Polymer	Chair mass (kg)	Reference flow (kg/FU)
HDPE (all types of feedstock)	2.40 <sup>(1)</sup>	2.40
PP (all types of feedstock)	2.29 <sup>(2)</sup>	2.29

<sup>(1)</sup> Calculated based on the mass of PP chairs, assuming an average material density equal to 908 kg/m<sup>3</sup> for PP and 950 kg/m<sup>3</sup> for HDPE.

<sup>(2)</sup> Mass of a real type of PP stacking chairs available on the market.

## 12.3 System boundary

In all scenarios, the system boundary was set in order to cover the most relevant stages and processes of the full product life cycle (cradle-to-grave perspective), as described below and depicted in Figures 12.1 to 12.6:

- *Feedstock Supply*<sup>206</sup> – covering extraction, transport and possible refining of crude oil and natural gas (fossil-based polymers), collection, transport and sorting of plastic waste (recycled polymers), crop cultivation (bio-based polymers), as well as transport of these feedstock sources to downstream conversion processes (e.g. naphtha cracking, polymer recycling, sugarcane fermentation);
- *Polymer Production*<sup>207</sup> – covering all the activities associated with the conversion of feedstock materials into the relevant monomer(s) and final polymer, including any transport among these activities and final transport of polymer granulate to downstream manufacturing processes;
- *Article Production*<sup>208</sup> – including chair manufacturing through injection moulding;
- *Distribution* – including transport of the chair from the manufacturing site to retailers and from these to final consumers;
- *End of Life* – covering collection, transport, recycling, incineration, or disposal of the chairs after use, including any avoided processes from virgin material or energy substitution.

A different nomenclature was applied for some life cycle stages compared to the default nomenclature specified in the Method (and in the PEF framework) to make it more relevant for the investigated supply chains and the project scope. Moreover, the default “Raw Material Acquisition and Pre-processing” stage was further split into two separate sub-stages (i.e. Feedstock Supply and Polymer Production), to allow disaggregating the impacts of feedstock supply from downstream conversion processes, and hence to better appreciate any differences among the use of different feedstock sources (in line with the project goal).

The Use Stage was excluded, as it can be reasonably assumed to involve no (or negligible) burdens, as well as to be identical across all the compared scenarios. Therefore, it can be omitted from a comparative assessment, as the one performed in this study. An exception may be represented by the possible release of (small) plastic particles to the environment, due to material degradation, in case of prolonged outdoor use. However, there is currently no evidence concerning this possibility, and any quantification would only be based on assumptions. Moreover, no impact assessment methods are currently available to capture the potential impact of plastics directly released into the environment (see Section 3.4). On the other hand, this aspect should be adequately addressed (as part of additional environmental information) by any company, producer or other supply chain actor undertaking a LCA study, especially if specific product information is directly available to them.

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 chairs, of the examined polymers and of plastics in general, as well as on their release and fate over the product life cycle. This is acknowledged as a limitation of this study, as additive production can account for a non-negligible portion of 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 (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 plastics routed to biological treatments or subject to in-situ degradation).

<sup>206</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

<sup>207</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

<sup>208</sup> Corresponding to the default stage of “Manufacturing” specified in the Method and in the PEF framework.

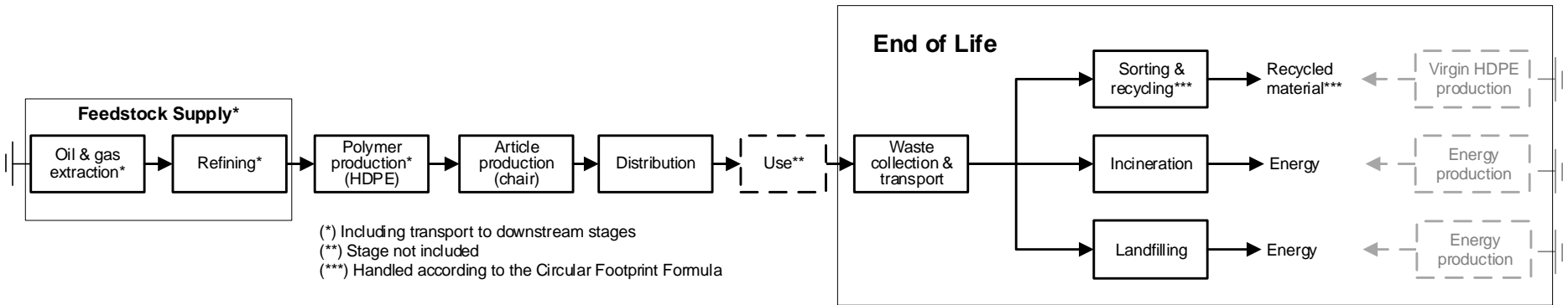


Figure 12.15 System boundary for fossil-based HDPE monobloc stacking chairs (Scenario 1).

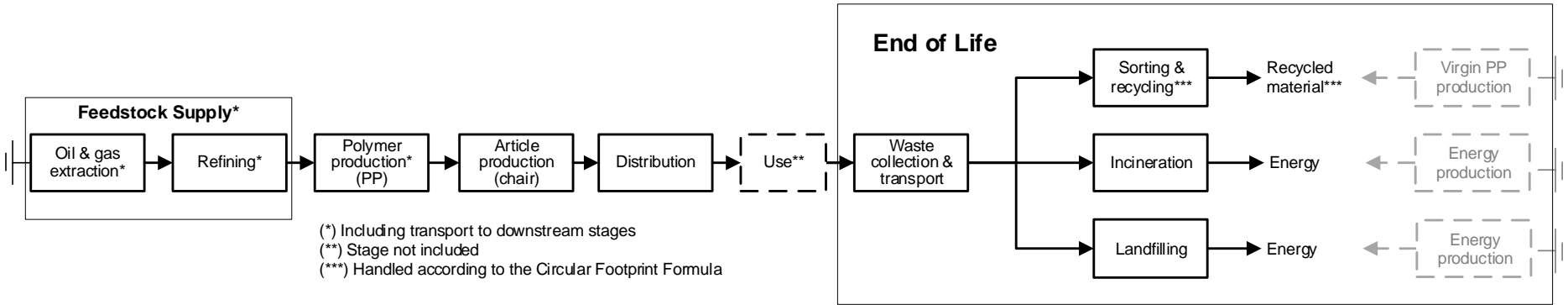
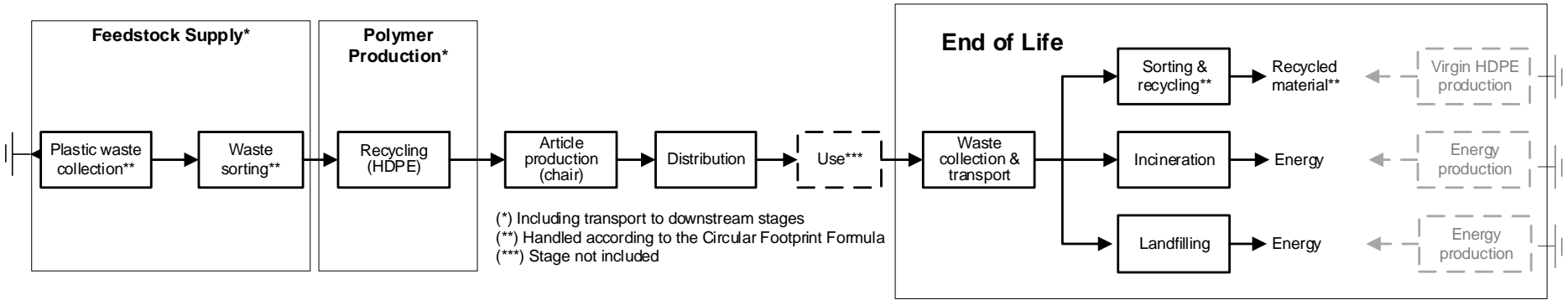
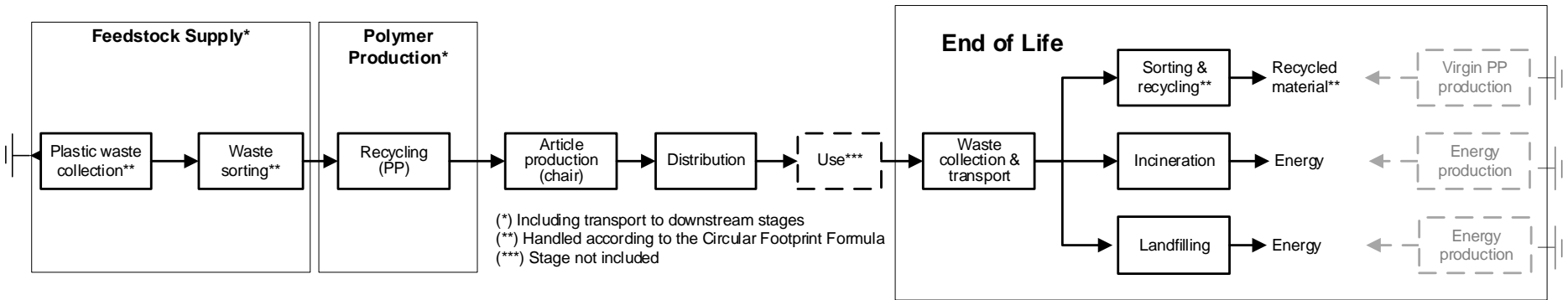


Figure 12.16 System boundary for fossil-based PP monobloc stacking chairs (Scenario 2).



**Figure 12.17** System boundary for 100% recycled HDPE monoblock stacking chairs (Scenario 3).



**Figure 12.18** System boundary for 100% recycled PP monobloc stacking chairs (Scenario 4).

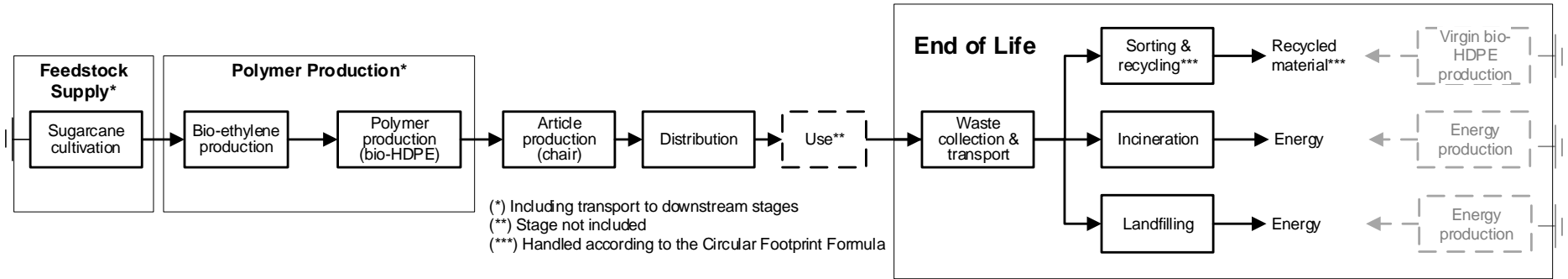


Figure 12.19 System boundary for bio-based HDPE monobloc stacking chairs (Scenario 5).

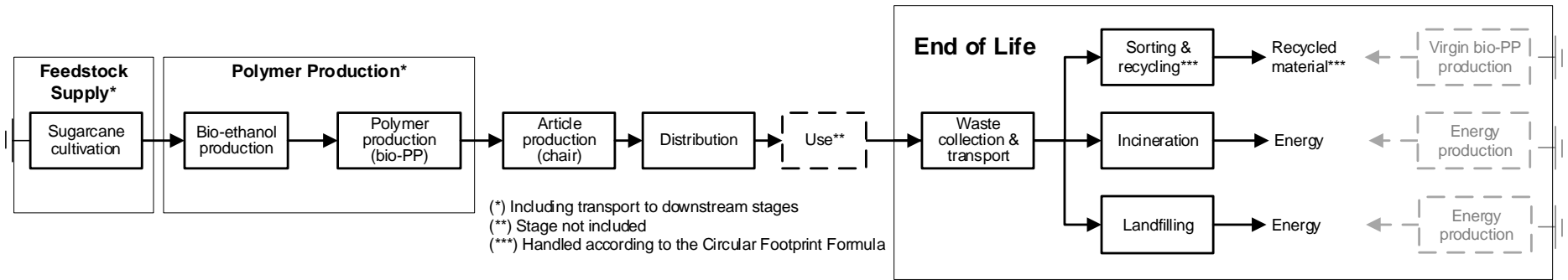


Figure 12.20 System boundary for bio-based PP monobloc stacking chairs (Scenario 6).

## **12.4 Life Cycle Inventory**

This section describes the overall approach for developing the Life Cycle Inventory of the analysed scenarios, along with the related assumptions and data sources. The description is separated by major lifecycle stages, in the following sub-sections (12.4.1 – 12.4.6). The list of processes, related data sources, and main modelling details are provided in Tables B.9.1 to B.9.6 in Annex B.9.

### **12.4.1 Feedstock Supply Stage**

#### **12.4.1.1 Fossil-based polymers**

For fossil-based polymers (PP and HDPE), the stage of Feedstock Supply includes the processes of crude-oil and natural gas extraction and transport, naphtha production in crude oil refineries, and its subsequent transport to downstream conversion processes (i.e. naphtha cracking). The modelling of these activities was carried out as described in Section 4.4.1.1, where the reader is referred to for further details on data sources, covered sub-processes and related environmental burdens, main modelling choices (e.g. allocation) and any adjustments performed in the implementation of the applied datasets.

#### **12.4.1.2 Recycled polymers**

For recycled polymers (R-PP 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. These processes were modelled as already described in Section 4.4.1.2 for the Beverage Bottle case study, and refer to collection and transport of separately collected plastic waste at the municipal level (in the absence of specific data for plastic waste collection from the different sectors it may come from), as well as to sorting of mixed plastic waste in dedicated facilities. Both processes were implemented in the lifecycle model according to the Circular Footprint Formula (CFF), which is the default approach to handle recycling situations in the Product Environmental Footprint (PEF) context. According to this approach, only 50% of the burdens from collection, transport, sorting and recycling were assigned to the recycled content in chairs ( $A=0.5$  for HDPE and PP used in unspecified applications), the rest being assigned to the system providing material for recycling. Further details on the implementation of the CFF are provided in Section 4.4.2.2 on recycled polymers production.

#### **12.4.1.3 Bio-based polymers**

For the bio-based polymers assessed (Bio-HDPE and Bio-PP), the stage of Feedstock Supply includes cultivation of sugarcane in Brazil, and its subsequent transport to further processing in the same country. Growing of Brazilian sugarcane (used as a feedstock for bio-propylene and bio-ethylene) was modelled through aggregated datasets from the GaBi database. Used in combination, these datasets depict a situation where 45% of sugarcane is manually harvested via the "slash and burn" practice, i.e. sugarcane residues (tops and leaves) are burned on standing plants before harvesting. This share was considered representative of the current situation. However, the slash and burn 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). Transport of harvested sugarcane to further processing in sugarcane mills for Ethanol 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).

### **12.4.2 Polymer Production Stage**

The Polymer Production stage covers the activities of feedstock processing into relevant intermediates and monomer(s), the polymerisation process, as well as any transport

among these activities and final transport of polymer granulate to the chair manufacturing site. The following subsections (12.4.2.1 – 12.4.2.4) describe how these activities have been modelled in the present case study.

#### **12.4.2.1 Fossil-based polymers**

For conventional, fossil-based polymers (HDPE and PP), the whole process chain from feedstock processing to polymerisation, through the production of intermediates and monomers, was modelled by means of aggregated, gate-to-gate datasets provided by Thinkstep. Inputs include combinations of crude oil, natural gas and naphtha, depending on the polymer. All conversion processes are assumed to take place in Europe, so that the datasets reflect EU average background conditions in terms of e.g. energy generation, material supply and transport. For both polymers, the main conversion process involved in the supply chain is steam cracking of naphtha and natural gas, delivering the monomers Ethylene and Propylene, along with Butadiene and other relevant intermediate flows (e.g. pyrolysis gas including mixtures of Benzene, Toluene and Xylenes). The inventory of this process is mainly based on industry data, completed, where necessary, by literature data. 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 co-products. The final polymerisation process of HDPE and PP is mainly modelled based on industry data from internationally adopted production processes, integrated with literature data when needed. No allocation is performed for this stage, being HDPE and PP the only outputs from the respective production process.

#### **12.4.2.2 Recycled polymers**

The production of recycled HDPE granulates out of sorted, post-consumer HDPE waste was modelled building upon the *ecoinvent* dataset “[*Europe without Switzerland*] Polyethylene production, high density, granulates, recycled” (no specific EF-compliant dataset is available for modelling purposes). However, the latter was updated according to the values and (where needed) flows, reported in a most recent and expanded version of the original data source (i.e. Franklin Associates, 2018, updating Franklin Associates, 2011) and determined as mass-weighted averages of data collected from several recycling facilities in the United States. Updated exchanges were then combined with EF background datasets for energy and material supply under EU-average conditions. The overall recycling efficiency is equal to 84%, with removed contaminants and processes waste being sent to incineration (which along with co-combustion in cement kilns is one of the two typical fate of plastic recycling residues with high calorific value; Rigamonti et al., 2014).

Data from the abovementioned source (i.e. Franklin Associates, 2018) were also used to develop a new inventory dataset related to the production of recycled PP granulate from sorted, post-consumer PP waste. Similarly to recycled HDPE resin, such data represent mass-weighted averages of values collected from several recycling plants in the United States, and for modelling purposes were used in combination with EF background datasets for energy and material supply under EU-average conditions. In this case, the overall recycling efficiency is equal to 85.5%, with removed contaminants and process waste still sent to incineration (consistently with the assumption in the HDPE recycling process described above, and the typical fate of plastic recycling residues).

According to the approach specified in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the recycling process were allocated to the recycled content in chairs ( $A = 0.5$  for HDPE and PP used in unspecified applications). However, the recycled content carries 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 life cycle providing the recycled material). Assuming a value of the  $Q_s/Q_p$  factor equal to 0.9 (similarly to the value recommended for HDPE and PP when used in packaging applications, in the absence of more specific

data) the allocated share of virgin production impacts is equal to 45% ( $A \times Q_s/Q_p = 0.5 \times 0.9 = 0.45$ ). Virgin polymer production burdens were modelled as described above (Section 12.4.2.1) for conventional, fossil-based HDPE and PP, and in Section 12.4.1.1 for the respective Feedstock Supply.

### **12.4.2.3 Bio-based polymers**

#### **12.4.2.3.1 Bio-based HDPE**

For bio-based HDPE production, no datasets are available from commercial databases or the set of EF-compliant datasets. All conversion processes 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 considers Brazil as a relevant geography. 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 bioethanol-based polymers investigated in this project (i.e. Bio-PET, Bio-LDPE and Bio-PP), and to better align to the hierarchy for the handling of multi-functionality adopted in the present 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 metals<sup>209</sup>. The specific amount of sugarcane required for Ethanol production is equal to 15 kg per kg of Ethanol.

Ethanol produced in Brazil was assumed to be transported to Europe for further conversion and polymerisation, consistently with the assumption performed in the aggregated, gate-to-gate datasets used to model the production of the other bioethanol-based polymers considered in this project. Transport is modelled according to the default transport scenario specified in the PEF context (and in the present 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 ethanol 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.com<sup>210</sup>, 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 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 producers of bio-based HDPE is located in Brazil. In this specific case, the whole process chain of conversion and polymerisation would take place in Brazil, with the final HDPE resin being eventually transported to Europe. The assumption performed in this study is thus slightly in disfavour of the Bio-HDPE supply chain, since nearly 2 kg of bioethanol

<sup>209</sup> Infrastructure processes related to the Ethanol 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 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 Freshwater ecotoxicity.

<sup>210</sup> Available at: <https://www.searates.com/services/distances-time/>



are required per kg of Bio-HDPE (see below), and hence a higher mass of material is transported per functional unit compared to the transport of the final HDPE.

The inventory for 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 report were combined with background EF datasets for energy generation, and *ecoinvent* background datasets for material production<sup>211</sup>. Reported transport activities were not implemented in the inventory, being transport of Ethanol (the main raw material) accounted for separately in the foreground inventory (i.e. the overall life cycle model), while transport of the other input materials is included in the datasets applied to model their supply. Beyond Ethylene, a small amount of naphtha is also obtained as a co-product, which is handled via direct substitution of naphtha from crude oil refinery (modelled as described for the “Feedstock Supply” stage in Section 4.4.1.1). The data source does not report the specific Ethanol requirement, which was thus determined as the average of the values reported in IEA-ETSAP & IRENA (2013; 1.74 kg Ethanol/kg Ethylene) and IfBB (2018; 2.08 kg Ethanol/kg Ethylene), corresponding to an estimated consumption of 1.91 kg Ethanol per kg of Ethylene.

The final polymerisation step of Ethylene to HDPE was modelled based on data from the most recent PlasticsEurope ecoprofile (PlasticsEurope, 2016), as implemented in the *ecoinvent* database. Indeed, inventory data limited to the polymerisation stage 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 this process across all the investigated HDPE-based scenarios (i.e. fossil, recycled, and bio-based HDPE chairs). 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 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 production<sup>212</sup>. No allocation or 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 (2017), i.e. 1.07 kg Ethylene/kg, representative of suspension polymerisation.

#### 12.4.2.3.2 Bio-based PP

The production of bio-based PP out of Brazilian sugarcane was modelled based on an aggregated dataset provided by Thinkstep. The dataset covers the steps of sugarcane processing to bioethanol in Brazil (fermentation and distillation), bioethanol transport to Europe via transoceanic ship, conversion to bio-Ethylene (via dehydration) and subsequently to Propylene (via dimerization and metathesis), as well as final polymerisation to PP. 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. For the remaining conversion and polymerisation processes, no allocation

<sup>211</sup> 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.

<sup>212</sup> 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 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] *Market for chemical, organic*), i.e. acetic acid, methanol, urea, vinyl acetate, ethylene dichloride, and formaldehyde). The respective shares were then equally subdivided among remaining chemicals.

nor substitution need to be applied. The inventory is mainly based on industry data from internationally adopted production processes, completed, where necessary, by literature data. The final (gas phase) polymerisation step specifically relies on secondary data. Background inventories from the GaBi database are used to model the burdens of the inventoried inputs and outputs.

#### **12.4.2.4 Transport of polymer granulate to the article production site**

Modelling of transport of polymer granulate from the polymerisation plant (inside or outside the EU), to the chairs manufacturing site in Europe, is based on the default transport scenario (distances and vehicle types) specified in the PEF context (and in the 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 HDPE and PP), the following routes were thus considered:

- (i) 130 km by lorry (total weight >32 t; Euro 4);
- (ii) 240 km by train (average freight); and
- (iii) 270 km by ship (barge).

For the imported share of fossil-based HDPE and PP (22% and 10%, respectively), a transoceanic ship transport was considered as the main transport route. The corresponding overall 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)<sup>213</sup>. Countries contributing to at least 90% of the overall import were considered in the calculation, for an overall distance equal to 7,982 km for fossil-based HDPE, and to 8,609 km for fossil-based PP (see Tables 12.4 and 12.5). Oceanic ship transport was complemented with road transport to and from the harbour in both the exporting and importing country (i.e. the EU). Road transport is made by lorry (total weight >32 t; Euro 4) along a distance of 1000 km. LCIs for all types of vehicles are available as EF-compliant datasets, which were used in the modelling.

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<sup>213</sup> Available at: <https://www.searates.com/services/distances-time/>

**Table 12.4.** Calculation of the overall average sea distance for imports of fossil-based HDPE to Europe.

Exporting country	Import (%) <sup>(1)</sup>	Import (% cum.)	Distance (km) <sup>(2)</sup>	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>				<b>7,982</b>

<sup>(1)</sup> Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

<sup>(2)</sup> 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 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.

**Table 12.5.** Calculation of the overall average sea distance for imports of fossil-based PP to Europe.

Exporting country	Import (%) <sup>(1)</sup>	Import (% cum.)	Distance (km) <sup>(2)</sup>	Weighted distance (km)
SAUDI ARABIA	41.3	41.3	8767.28	3622
KOREA, REPUBLIC OF (SOUTH KOREA)	13.5	54.8	16702.31	2259
INDIA	7.15	62.0	10267.28	734
RUSSIAN FEDERATION (RUSSIA)	6.75	68.7	5574.44	376
BRAZIL	6.63	75.4	10107.97	670
ISRAEL (GAZA and JERICOHO- >1994)	4.86	80.2	2987.48	145
SOUTH AFRICA (incl. NA - >1989)	4.26	84.5	12160.92	518
EGYPT	3.20	87.7	3212.14	103
UNITED STATES	2.99	90.7	6061.9	181
Other countries	9.32	100	-	-
Overall weighted distance				8,609

<sup>(1)</sup> Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

<sup>(2)</sup> 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 Middle-East and Asia were determined considering Marseille as destination port in Europe. For imports from other countries (in the case of PP Russia, Brazil, South Africa and United States), Rotterdam was considered as destination port.

### 12.4.3 Article Production Stage

Regardless of the feedstock used, manufacturing of both HDPE and PP chairs was assumed to take place via injection moulding, which is one of the most commonly applied process to convert raw polymer granules to plastic goods. In the injection moulding process, melted or plasticized plastic material is injected by force into a clamped mould cavity (consisting of a single cavity or a number of similar or dissimilar cavities), where it is finally shaped into the desired object (Rosato et al., 2004), chairs in this case.

The burdens of the overall conversion process were modelled through the aggregated, EF dataset "[EU-28+EFTA] Injection moulding; plastic injection moulding | production mix, at plant | for PP, HDPE and PE", which accounts for a 97.1% conversion efficiency. Process losses (i.e. 2.9% of the polymer input) were assumed to be entirely recycled in external facilities via re-granulation into new polymer pellets, ultimately replacing virgin granules of the same material. Therefore, recycled HDPE granules were assumed to replace virgin, fossil-based HDPE granules, being the estimated share of bio-based HDPE on the market currently very low (i.e. 0.2%). Similarly, recycled PP granules were assumed to replace virgin, fossil-based PP granules, since no bio-based PP was reported to be available on the market at the time of developing this analysis (EUBP, 2019). The recycling process and the resulting virgin material substitution were modelled based on the same data as End of Life recycling of post-consumer HDPE and PP chairs, in the absence of more specific data for recycling of pre-consumer, industrial scraps. For further detail on the modelling of recycling and avoided virgin material production (including the implementation of the Circular Footprint Formula), the reader is hence referred to Section 12.4.5.3 on End of Life modelling.

#### **12.4.4 Distribution Stage**

The transport of chairs from the manufacturing site to the final client was modelled based on the default transport scenario specified in the PEF context (and in this method) for the route factory → retail → final client. The following routes were thus considered:

- (i) 1200 km by articulated lorry (total weight >32 t; Euro 4) from factory to retailers;
- (ii) 5 km by passenger car for 62% of the roundtrips from retailers to final clients;
- (iii) 5 km by van for 5% of the roundtrips from retailers to final clients; and
- (iv) no burdens assigned to 33% of the roundtrips from retailers to final clients (assumed to take place with no motorised vehicles).

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

#### **12.4.5 End of Life Stage**

This section describes the modelling of the End of Life Stage of chairs. In particular, Section 12.4.5.1 focuses on the definition of the EU-average End of Life scenario, which is considered as base case for the calculation of the potential impacts of the compared LCA scenarios. The remaining sections (12.4.5.2 – 12.4.5.5) address the modelling of waste collection and transport, and of the different End of Life options explored in the study. Finally, Section 12.4.5.6 provides case-specific details on the estimate of the potential contribution of the chairs to macro-plastic formation at the End of Life (including product litter) and micro-plastic generation throughout the supply chain.

##### **12.4.5.1 End of Life scenario**

Concerning the End of Life of chairs, no specific and representative data were available on the share in which the different viable End of Life options (including mechanical recycling, incineration and landfilling) are currently applied. Therefore, to define the EU-average End of Life scenario, the rates estimated for total plastic waste at the EU level were applied, i.e. 30% mechanical recycling (after separate collection and sorting), 39% incineration and 31% landfilling (EC, 2018). These rates were identically applied to all scenarios, as they are not affected by the feedstock used for polymer production, but rather by material properties, which are the same in all scenarios.

##### **12.4.5.2 Modelling of waste collection and transport**

After use by citizens or other users, waste chairs are (separately) collected and transported to the different considered End of Life options. These activities were modelled according to the pathways, vehicle types and distances reported in Rigamonti et al. (2013) for separately collected plastic waste at the municipal level, and for residual municipal waste sent to incineration or landfilling. These data refer to a region with a well-developed municipal 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 the considered collection pathways and data may be more representative of packaging waste or smaller (plastic) articles, they can be reasonably extended also to chairs discarded by municipal users. Moreover, since the modelling of collection and transport was implemented consistently across all the investigated scenarios, their comparison is not affected by these possible approximations and assumptions (but only their “absolute” impact is). More information on the main assumptions and modelling details of the considered collection pathways is available in Section 4.4.5.2. Note that when EF incineration and landfilling datasets were used, no transport of collected waste was separately modelled, since the respective burdens are already accounted for in such datasets. In this case, only collection was modelled, according to the approach described above.

### 12.4.5.3 Modelling of sorting and recycling

Before recycling, separately collected post-consumer chairs were assumed to be firstly sorted in specific facilities, similarly to other waste plastic articles from municipal collection. 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. HDPE and PP). Additional sorting of homogeneous polymer streams by colour may be performed, directly at sorting facilities or also before recovery at recycling plants.

An average life cycle inventory of mixed plastic waste sorting has been developed in Franklin Associates (2018), based on input/output data from different dual-stream and single-stream sorting facilities in the United States. This inventory has been considered as a reference for the modelling of the sorting process, complementing the reported input and output data with background EF datasets representative of EU-average conditions. In the implementation, a 100% sorting efficiency was assumed for the chair waste stream, which entirely consists of one of the targeted materials for recycling. Moreover, no burdens from the treatment of any impurities sorted out as rejects were assigned to the chair waste stream, to avoid falsely “punishing” them with burdens from unrelated product waste (and mostly coming from incorrect citizen behaviour). Therefore, the modelled sorting process only accounted for the material and energy inputs required to carry out this waste treatment activity. The final inventory of the sorting process applied in the modelling is summarised in Section 4.4.5.3, Table 4.9.

For mechanical recycling of both HDPE and PP chairs, no specific datasets are available in the PEF context. Hence, for HDPE-based chair recycling, the *ecoinvent* dataset “[Europe without Switzerland] Polyethylene production, high density, granulate, recycled” was used as a basis for modelling purposes, in combination with EF background datasets for energy and material supply under EU-average conditions. Since a most recent and expanded version of the original inventory data source is available (i.e. Franklin Associates, 2018, updating Franklin Associates, 2011), the dataset was adjusted according to the updated exchange values and, if needed, exchange types, reported in the mentioned 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 being sent to incineration along with process waste. This is in line with the typical fate of recycling residues, which due to their high calorific value are normally sent to incineration or co-combustion in cement kilns (Rigamonti et al., 2014).

Similarly to HDPE chairs, a new dataset was developed for mechanical recycling of PP-based chairs, based on foreground inventory data available as well in Franklin Associates (2018)<sup>214</sup>. These still represent mass-weighted average values of point data collected from several US recycling facilities, which for modelling purposed were here combined with EF background datasets for energy and material supply (under EU-average conditions). The inventory accounts for an overall recycling efficiency equal to 85.5%, with removed contaminants and process waste being sent to incineration, consistently with the typical fate of plastic recycling residues, and the assumption performed in the HDPE recycling process described above.

Recycled polymer granulate was assumed to replace virgin granulate of the same material, whose primary production burdens were credited to the system. For polymers having both a fossil-based and a bio-based alternative available on the market as of today (i.e. only HDPE in this case study) 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 only 0.2% for HDPE (which is for

<sup>214</sup> An aggregated EF-compliant dataset is also available for PP recycling in the US (based on an older version of the same inventory data source, i.e. Franklin Associates, 2011). However, since this could not be adjusted to reflect EU background conditions (due to its aggregated nature) and most recent data were available, a new disaggregated dataset was developed.

99.8% fossil-based)<sup>215</sup>. To account for the lower overall quality of recycled polymers compared to the replaced virgin polymers, a substitution ratio equal to 0.9 was considered for HDPE and PP, according to default values specified in the PEF context for such materials when used in packaging applications (in the absence of more specific data). To model the burdens of avoided primary production, the same datasets (or combination of datasets) used for the modelling of upstream production of the relevant polymers were applied (as described in Sections 12.4.2, and 12.4.1 for the related feedstock). This was made for consistency reasons, and to avoid possible distortions by applying different datasets from other sources.

According to the approach adopted in the PEF context to model recycling situations (Circular Footprint Formula), only 50% of the burdens of the sorting and recycling processes were allocated to the system ( $A = 0.5$  for both PE and PP used in unspecified applications). Similarly, only 50% of the benefits from avoided virgin material production were assigned to the system itself.

#### **12.4.5.4 Modelling of incineration**

For the incineration process of conventional fossil-based polymers (i.e. HDPE and PP), aggregated, material-specific, EF-compliant datasets are available, and were used to model the treatment of chairs made of these materials in a municipal waste incineration plant. Similarly, for bio-based HDPE and bio-based PP chairs, partially aggregated, material-specific inventories from the GaBi database were applied (no EF datasets are available for these polymers).

All the selected datasets are developed based on a waste-specific incineration model, which has been described more in detail in Section 4.4.5.4. 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 general approach to handle energy recovery situations adopted in the EF context and in this method, the product system generating the waste material sent to incineration (i.e. the chair life cycle, in this case) takes the full burdens from the incineration process. However, it is credited for 100% of the benefits from avoided production of conventional energy (electricity and heat) replaced by energy recovered from waste. While in EF incineration datasets these credits are already accounted for in the aggregated inventory, for GaBi datasets they were added to the main process inventory. Therefore, the EU residual electricity grid mix (as modelled in the EF 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 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 includes 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. Also, thermal energy from waste (11%) was excluded, as according to the approach described above for the handling of energy recovery situations, 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).

<sup>215</sup> The share of bio-based HDPE was estimated based on the global production capacity of bio-based PE in 2018 (European Bioplastic, 2019; 200 kt) and of PE as a whole in 2016 (PlasticsInsight, 2019; 103 Mt).

#### 12.4.5.5 Modelling of landfilling

Landfilling of chairs made of conventional non-biodegradable polymers (i.e. fossil-based HDPE and PP, both virgin and recycled) was modelled based on a common, aggregated EF dataset representing disposal of plastic waste in a managed, municipal solid waste landfill ([EU-28+EFTA] Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific sites)). The underlying inventory is material-specific, but refers to the average chemical composition and degradability values of generic plastic waste, rather than to those of the specific polymer. 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<sup>216</sup>. 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 emissions to the environment over a 100 year time horizon. Emissions occurring beyond 100 years are not accounted for in the model. Landfill gas generation is calculated based on the organic carbon content in the waste and the respective degradation rate assumed over 100 years (not reported), while, 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 rate of landfill gas, energy efficiencies of engines, collection rate of leachate and respective treatment efficiencies). Further details on how these parameters were set to reflect the EU-average situation are provided in Section 4.4.5.5.

The same dataset described above for landfilling of generic plastic waste was also applied as a proxy for landfilling of chairs made of “drop-in”, non-biodegradable, bio-based polymers, i.e. bio-based HDPE and bio-based PP. Compared to the original dataset, emissions of CO<sub>2</sub> and CH<sub>4</sub> 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 amounts of fossil CO<sub>2</sub> and CH<sub>4</sub> reported as released to air were converted to biogenic emissions. This approximation is considered acceptable, as reported CO<sub>2</sub> and CH<sub>4</sub> emissions are modest, and overall amounting to only 1.4% of the carbon content in the landfilled plastic material, despite they also include the contribution of emissions from background processes. This can be partly explained by the generally low degradation (and mineralisation) of conventional, non-biodegradable polymers in landfill, which is typically assumed to be in the range of 1% over 100 years (Doka, 2009b). On the other hand, it is acknowledged that this approach is in favour of the two bio-based polymers (i.e. Bio-HDPE and Bio-PP).

In line with the time horizon applied for landfill emission modelling in the selected dataset, (biogenic) carbon in polymers that is not degraded (mineralised) after 100 years from deposition, was considered to be never released from the landfill body. As a base case, this missed release of atmospheric carbon taken up during biomass growth is not reflected in Climate Change LCIA results for bio-based HDPE and PP pots, since characterisation factors for biogenic CO<sub>2</sub> flows have been set to zero (fully conforming to prescriptions in the PEF method). However, to show the relevance of this choice on the overall results, the Climate Change impact of the two bio-based alternatives was also calculated by acknowledging the effects of non-released biogenic carbon. For this purpose, a specific CO<sub>2</sub> uptake was modelled in the inventory, expressing the net amount of CO<sub>2</sub> taken up during biomass growth and not released (as CO<sub>2</sub> or CH<sub>4</sub>) during polymer

<sup>216</sup> While more polymer-specific landfilling datasets are available from other databases (i.e. *ecoinvent*), the mentioned EF dataset for landfilling of generic plastic was selected, as specifically referring to EU as the reference geography (in contrast to available polymer-specific datasets), and to keep consistency with the overall dataset selection “hierarchy” followed in this project (Section 3.5).



degradation in landfill. The uptake was calculated based on the biogenic carbon content in the polymer, and assuming a mineralisation rate over 100 years equal to 1% for both polymers (Doka, 2009b). While the mineralisation rate assumed in the dataset is not reported explicitly, it seems to be in line with this assumption (according to the figures reported above regarding the inventoried carbon emissions). Considering a biogenic carbon content equal to 81.9% for both Bio-HDPE and Bio-PP, a carbon uptake equal to 2.97 kg CO<sub>2</sub>/kg bio-based polymer was calculated. This uptake was then characterised with a factor of -1 kg CO<sub>2</sub> eq. per kg CO<sub>2</sub>-C not degraded.

#### 12.4.5.6 Contribution to macro- and micro-plastics generation (including product litter)

The contribution to macro- and micro-plastics loss and release of the analysed chairs scenarios was estimated according to the framework outlined in Annex B of the Method (Report I), and summarised in Section 4.4.5.6 of this report in terms of underlying quantification methods and approaches. In this section, the case-specific parameters considered to apply the latter to the assessed chairs LCA scenarios are reported.

##### 12.4.5.6.1 PLP method

Regarding the estimate of the loss and release of macro-plastics at the End of Life stage (from product littering and waste mismanagement), the product-specific parameters reported in Table 12.6 were considered to apply the first level of the framework ("PLP method") to chairs (derived from Peano et al., 2020). Note that for the scenarios based on HDPE release rates are lower due to a higher economic value of the material.

**Table 12.6.** Case-specific parameters of the *PLP method* applied to the chairs LCA scenarios.

Parameter <sup>(1)</sup>	Value (HDPE, R-HDPE, Bio-HDPE)	Value (PP, R-PP, Bio-PP)
Littering rate (LR <sub>lit</sub> ) (%)	0	0
Release rate to ocean (Rel <sub>ocean</sub> ) (%)	1	5
Release rate to the terrestrial environment (Rel <sub>terenv</sub> ) (%)	1	95

<sup>(1)</sup> For details on the meaning of each parameter, refer to Annex B of the Method (Report I).

No case-specific parameters are required to calculate the supply-chain loss and release of micro-plastics to ocean and to the terrestrial environment by applying the PLP method. Indeed, the contribution of the considered micro-plastics sources (pellet loss and tire abrasion) depends on the magnitude of the relevant life cycle processes (i.e. polymer production and road transport) in the specific chairs scenario (which in turns depend on the reference flow<sup>217</sup> of each scenario). However, the quantification is made by means of default parameters that are not affected by the type of product, polymer or feedstock source (as reported in Annex B of the Method (Report I)).

##### 12.4.5.6.2 Expanded PLP method

The parameters required to calculate the micro-plastics loss and release from marine coatings and road markings are not case-specific, and are reported in Annex B of the Method (Report I). Similar to the other considered micro-plastics sources, the contribution of these additional sources depends on the intensity of relevant (transport) processes in the specific product inventory, by means of the above-mentioned case-unspecific parameters.

As for the calculation of the additional contribution of plastic waste exported for recycling to the loss and release of macro-plastics from waste mismanagement, the case-unspecific parameters described in Annex B of the Method (Report I) were applied, in

<sup>217</sup> The reference flow is the mass of chair material required to fulfil the functional unit.

combination with the EU-average recycling rates estimated for each type of chairs  
(Section 12.4.5.1), i.e. 30% for all scenarios.

#### **12.4.6 Calculation of iLUC impacts**

As a base case, the contribution of Indirect Land Use Change (iLUC) to the potential  
Climate Change impact of bio-based chairs alternatives was calculated according to the  
approach outlined in Section 4.4.16.4 of the Method (Report I).

In order to apply (recalculated) iLUC GHG emission factors from the EU 2015/1513  
Directive (EC, 2015), the specific land demand for the production of relevant crops  
(m<sup>2</sup>\*year / kg crop) was calculated first. The calculation was based on the aggregated  
amount of arable and agriculture land occupation flows reported, for the relevant country  
(e.g. Brazil for sugarcane), in the dataset used to model the production of the specific  
crop. If the geography of such flows was not specified, all the arable and agricultural land  
occupation flows reported in the dataset were aggregated. These estimates were checked  
against values of land demand calculated based on 5-years average crop yields from  
FAOSTAT (FAO, 2019), which were found to be generally aligned with the former  
(absolute variation between 3% and 20%). Hence, the values estimated based on land  
occupation flows were applied, for consistency with the overall LCI modelling of  
scenarios.

Specific land demand for crop production was then converted into a demand per  
functional unit, based on the specific consumption of crop for polymer production (kg  
crop/kg polymer, consistently with the values applied in the LCI modelling) and the  
amount of polymer needed to fulfil the functional unit (reference flow) in the relevant  
scenario. The iLUC contribution to the Climate Change impact was finally calculated by  
applying to the latter the recalculated GHG emission factors from the Directive. The  
described calculation steps to estimate the iLUC contribution to the potential Climate  
Change impact are summarised in Table 12.7.

**Table 12.7.** Calculation of the iLUC contribution to the potential Climate Change impact of monobloc stacking chairs LCA scenarios relying on bio-based polymers.

Scenario / Polymer	Feedstock	Land demand for crop production <sup>(1)</sup> [m <sup>2</sup> *y/kg <sub>crop</sub> ]	Crop demand for polymer production [kg <sub>crop</sub> /kg <sub>polymer</sub> ]	Polymer demand per F.U. [kg <sub>polymer</sub> /FU]	iLUC GHG emission factor [kg CO <sub>2</sub> eq./ (m <sup>2</sup> *y)]	iLUC climate change impact [kg CO <sub>2</sub> eq./FU]
S5 – Bio-HDPE	Sugarcane (BR)	0.117 (0.135)	28.6	2.47	0.176	1.46
S6 – Bio-PP	Sugarcane (BR)	0.117 (0.135)	25.7	2.36	0.176	1.25

<sup>(1)</sup> Based on arable and agriculture land occupation exchanges reported in the dataset applied to the modelling of the specific crop. Values in parenthesis refer to land demand calculated based on crop yield data from FAOSTAT (5-years average), and are reported as a reference.

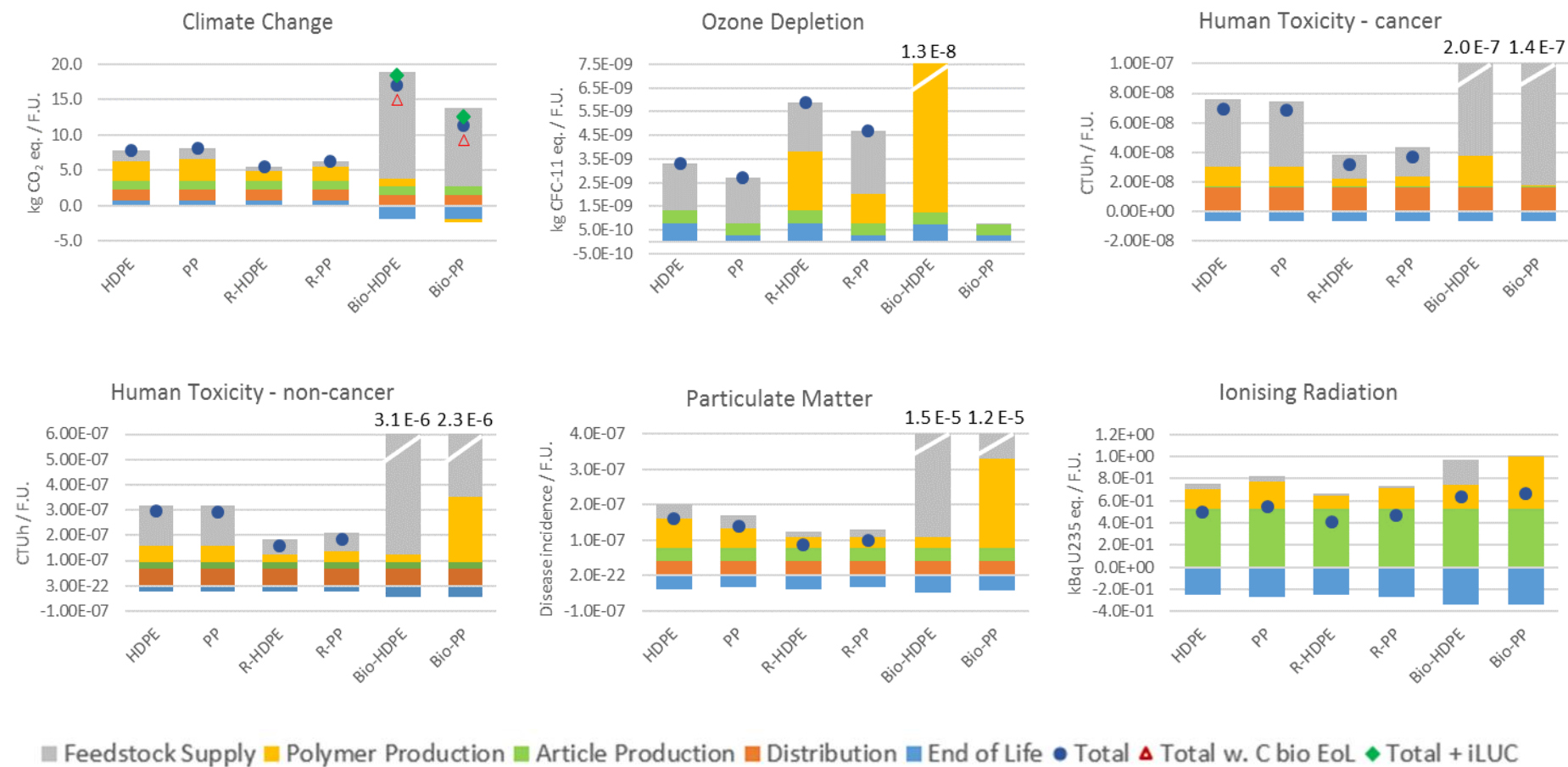
## 12.5 Life Cycle Impact Assessment results

The characterised potential impacts of the examined scenarios (with and without the iLUC contribution to Climate Change) are reported in Figures 12.7 to 12.9. These also show the breakdown of contributions from the main lifecycle stages, which include:

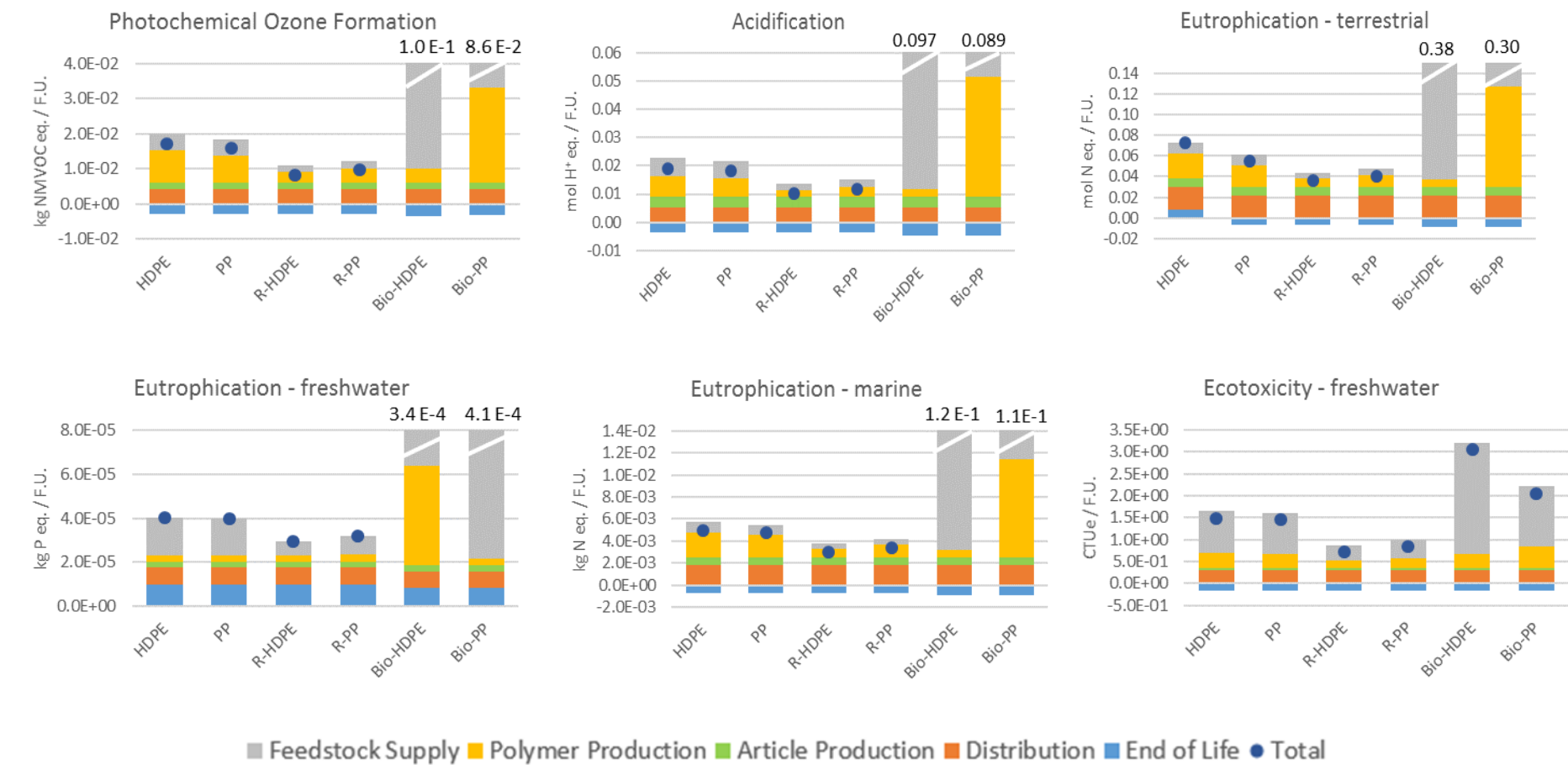
- i) *Feedstock Supply*, i.e. oil/natural gas extraction, transport and possible refining, transport of naphtha from refinery to downstream users; collection, transport and sorting of post-consumer plastic waste (recycled polymers); or crop cultivation and transport to further processing (bio-based polymers);
- ii) *Polymer Production*, i.e. all gate-to-gate activities from feedstock processing to monomer/intermediate production and polymerisation, including any transport among these, as well as transport of polymer granulate to the chairs manufacturing site;
- iii) *Article production*, i.e. conversion of the polymer into the finished article (i.e. injection moulding);
- iv) *Distribution*, i.e. transport, of chairs from manufacturing site to the final user; and
- v) *End of Life*, i.e. waste chair collection, transport and treatment or disposal, as well as any avoided processes from downstream displacement of virgin materials and energy.

Figure 12.10 shows the contribution of each sub-category to the Climate Change impact (i.e. from fossil, biogenic and land use/land transformation GHG emissions), while the overall characterised, normalised and weighted impact assessment results are available in Annex B.9. The latter are not reported in this section to keep readability of the document, despite this is not in alignment with the position suggested in the LCA report template provided in the Method (Report I).

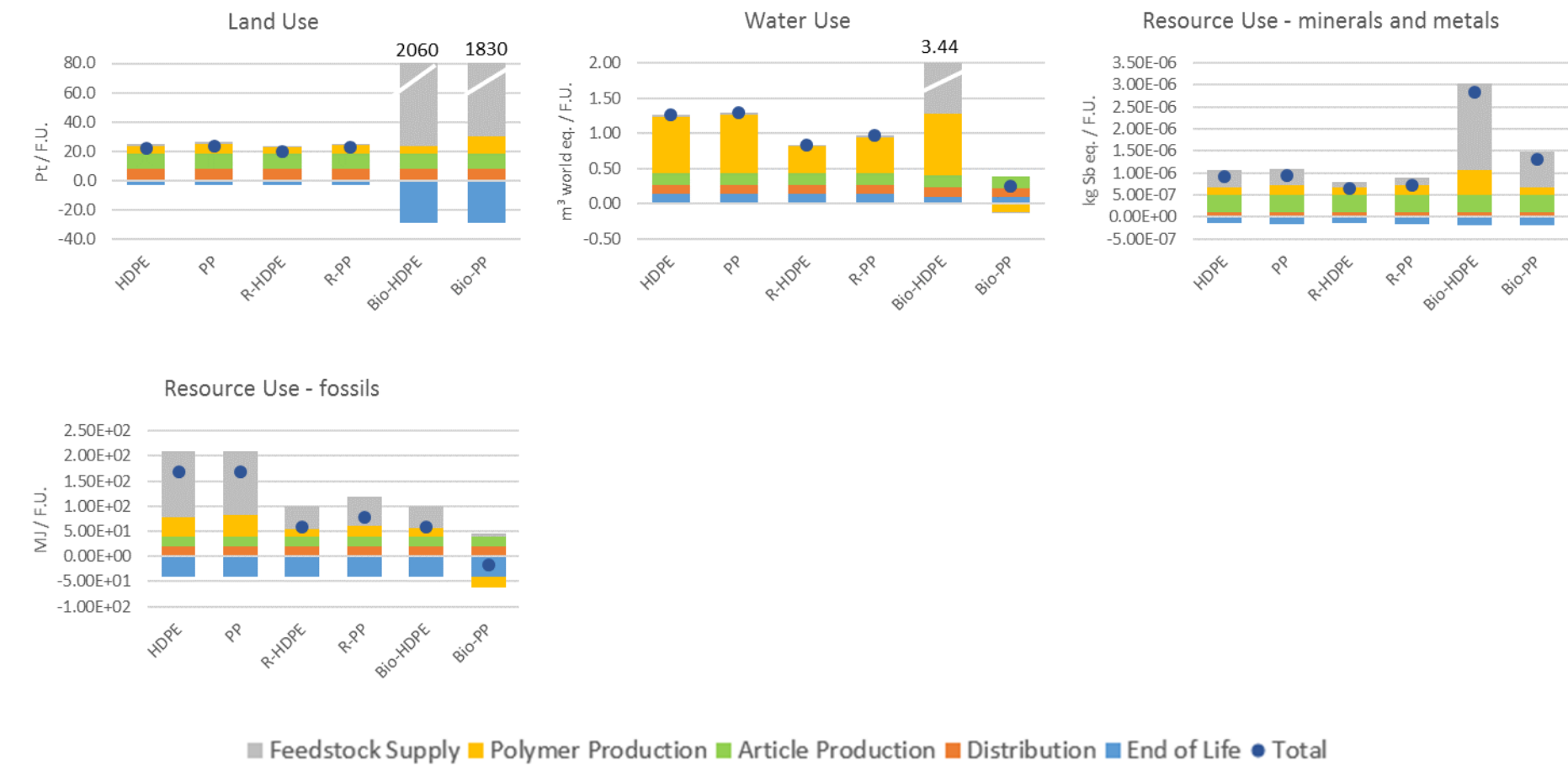
Note that scenario impacts presented in Figures 12.7 to 12.9 refer to the EU-average End of Life scenario (as described in Section 12.4.5.1), and represent the net impacts from waste management activities, resulting from the balance between real burdens and benefits (if any). Potential impacts calculated assuming 100% of chairs being routed to each viable End of Life option are presented in Figures 12.11 to 12.13. We warn that this analysis should not be interpreted as a direct LCA comparison of alternative End of Life management options for chairs, because a product-based perspective is considered, and allocations among consecutive product life cycles are applied to the recycling pathway. Conversely, in a typical waste management LCA that compares alternative End of Life options (i.e. with a functional unit of 1 tonne of waste managed), the recycling pathway would be assigned the full burdens and benefits it involves, as there is no need to break the system mass flows between the system recycling and the one using the recycled feedstock (i.e. system perspective; no allocation needed).



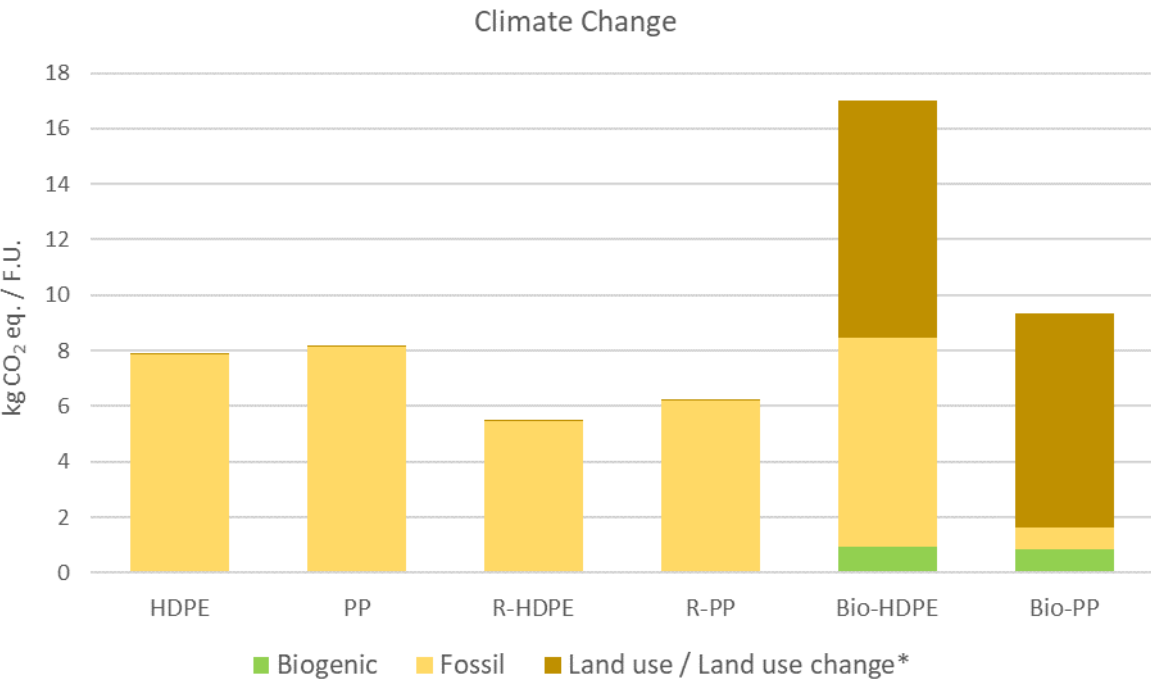
**Figure 12.7.** Potential impact of monobloc stacking chairs LCA scenarios for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity – non-cancer, Particulate Matter and Ionising Radiation. Note that in some impact categories a part of the results is out of scale and is curtailed. Climate Change impacts denoted with “C bio EoL” accounts for the contribution of biogenic carbon not released after 100 years from landfilling of bio-based chairs.



**Figure 12.8.** Potential impact of monobloc stacking chairs LCA scenarios for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater. Note that in some impact categories a part of the results is out of scale and is curtailed.

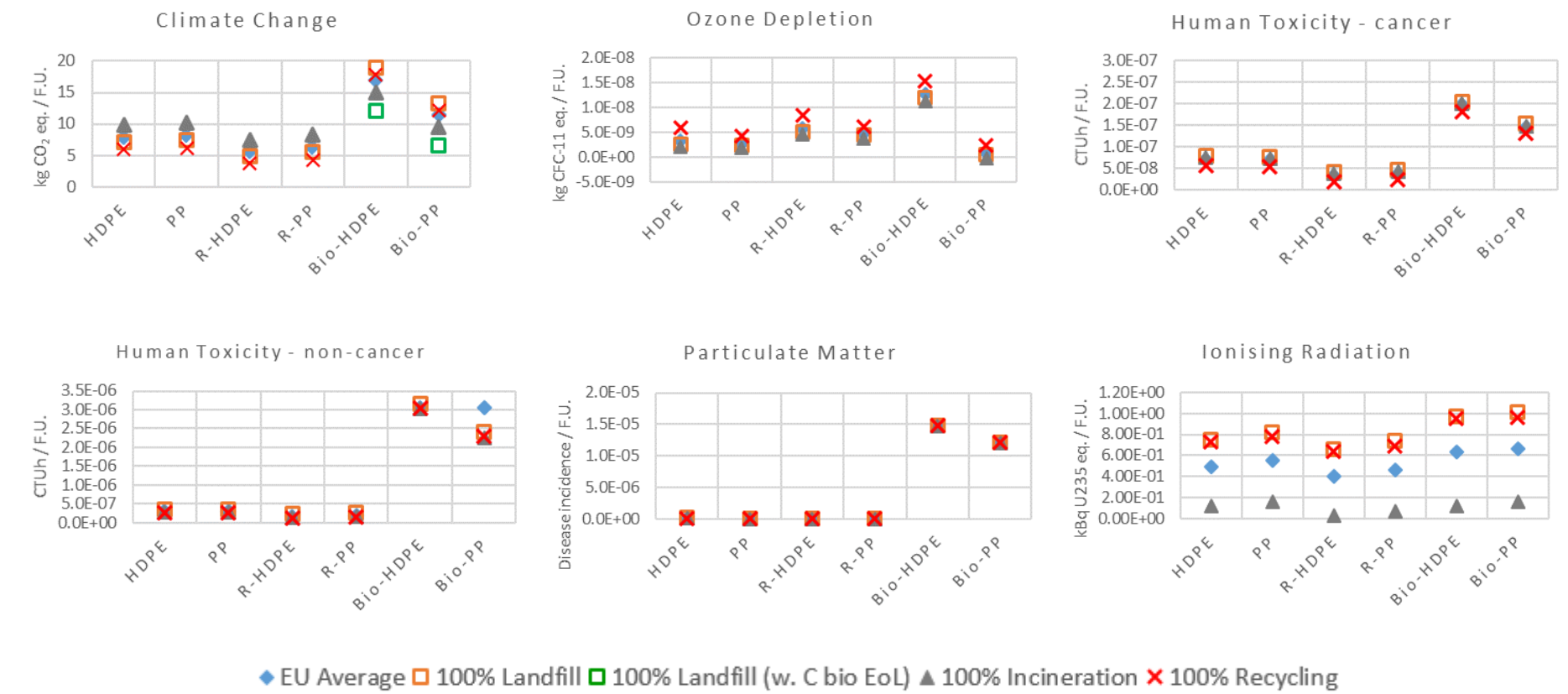


1 **Figure 12.9.** Potential impact of monobloc stacking chairs LCA scenarios for different End of Life options, for the categories of Land Use, Water Use,  
2 Resource Use - minerals and metals, Resource Use – fossils. Note that in some impact categories a part of the results is out of scale and is curtailed.  
3

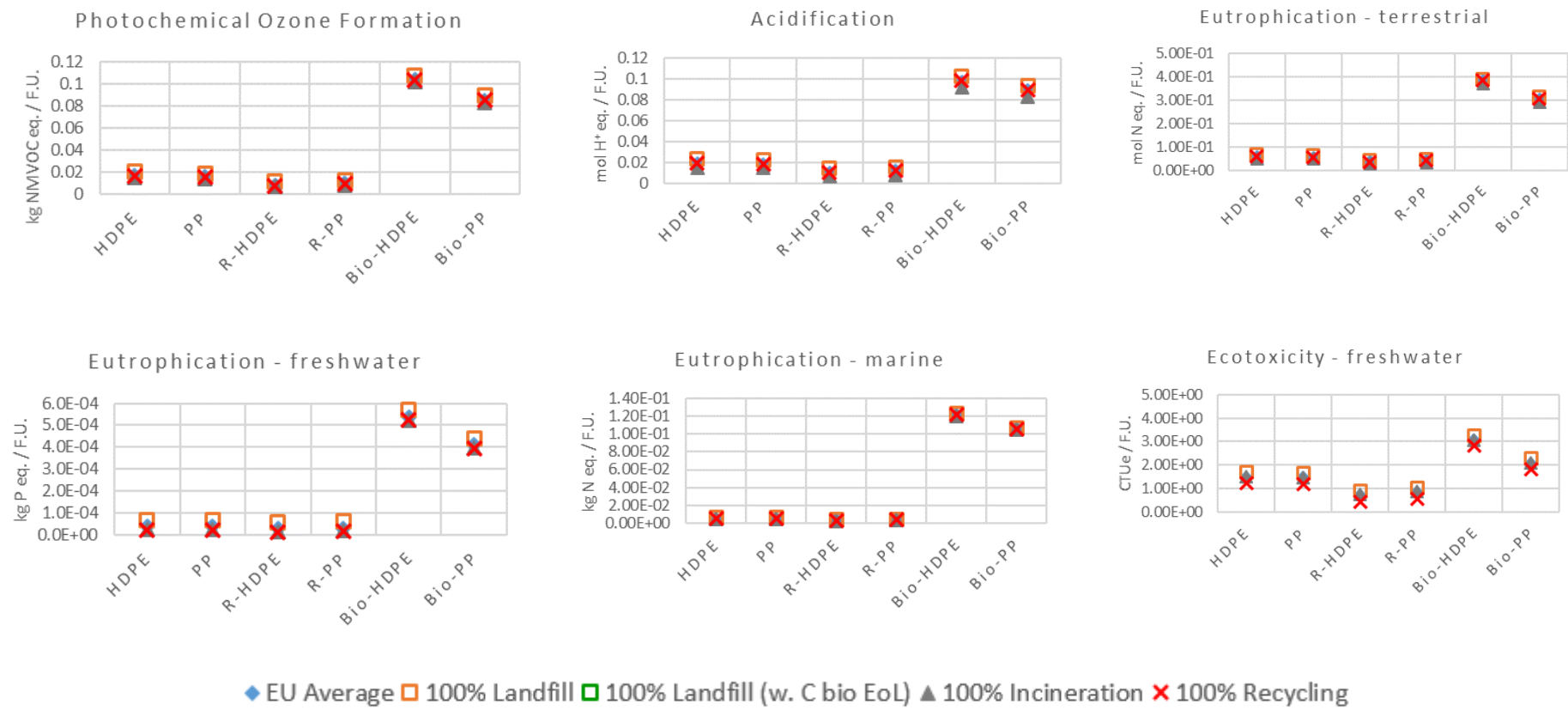


**Figure 12.10.** Contribution of fossil, biogenic, and land use/land transformation GHG emissions to the total Climate Change impact of monobloc stacking chairs LCA scenarios (note that the contribution of iLUC to land transformation emissions is excluded).

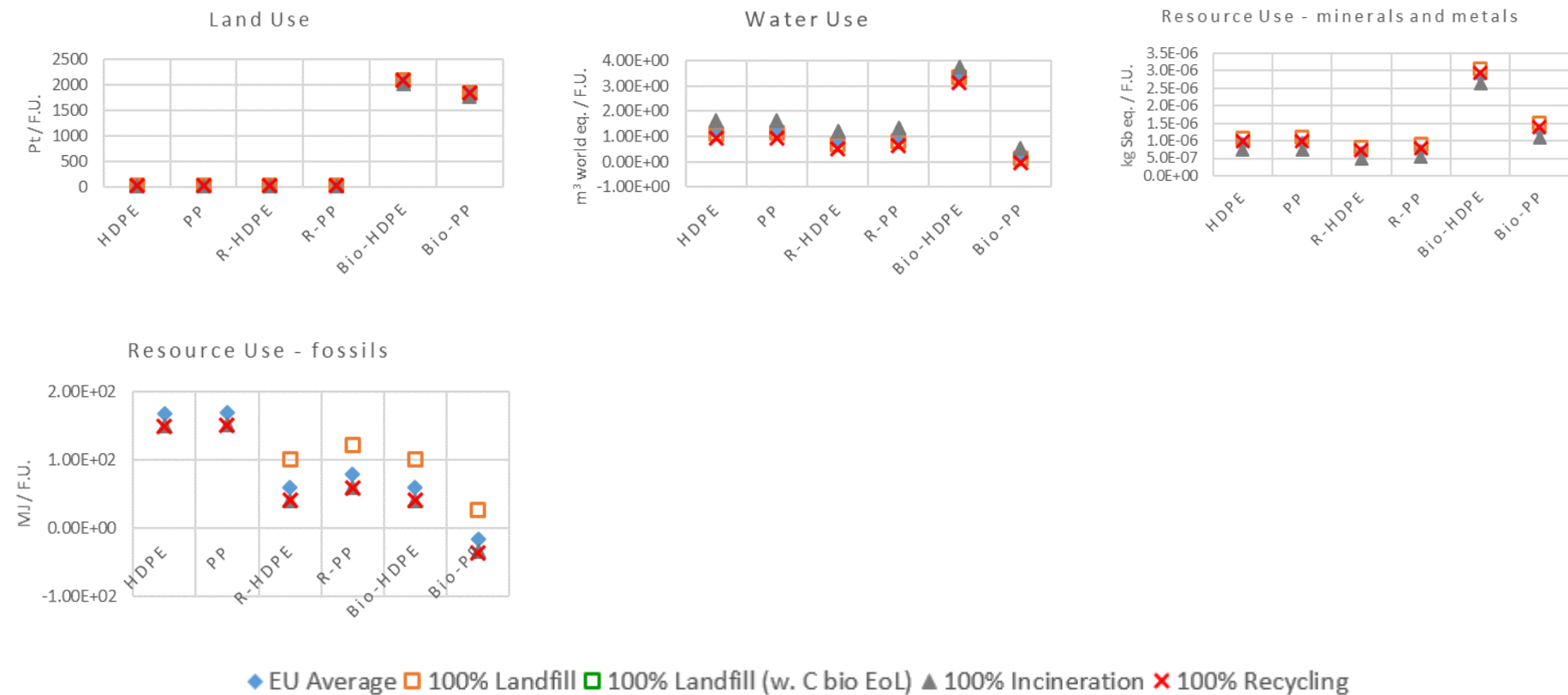




**Figure 12.11.** Potential impact of monobloc stacking chairs LCA scenarios for different End of Life options, for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity - non-cancer, Particulate Matter and Ionising Radiation. The Climate Change impact of 100% landfilling scenarios denoted with "C bio EoL" accounts for the contribution of biogenic carbon not released after 100 years from landfill deposition of bio-based chairs.



**Figure 12.12.** Potential impact of monobloc stacking chairs LCA scenarios for different End of Life options, for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater.



**Figure 12.13.** Potential impact of monobloc stacking chairs LCA scenarios for different End of Life options, for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils.

## 12.6 Additional Environmental Information

This section presents additional environmental impacts or aspects going beyond the default impact categories considered in the present method, which reflect official categories adopted in the PEF context. Additional environmental issues addressed include the impact from indirect Land Use Change (iLUC) on Climate Change, potential Biodiversity impacts, as well as the contribution of the investigated chairs scenarios to macro-plastics formation at End of Life (including product litter) and to micro-plastics generation throughout the supply chain.

### 12.6.1 iLUC impact

Table 12.8 presents the estimated contribution of GHG emissions from iLUC induced by bio-based feedstock supply to the potential Climate Change impact of bio-based chairs.

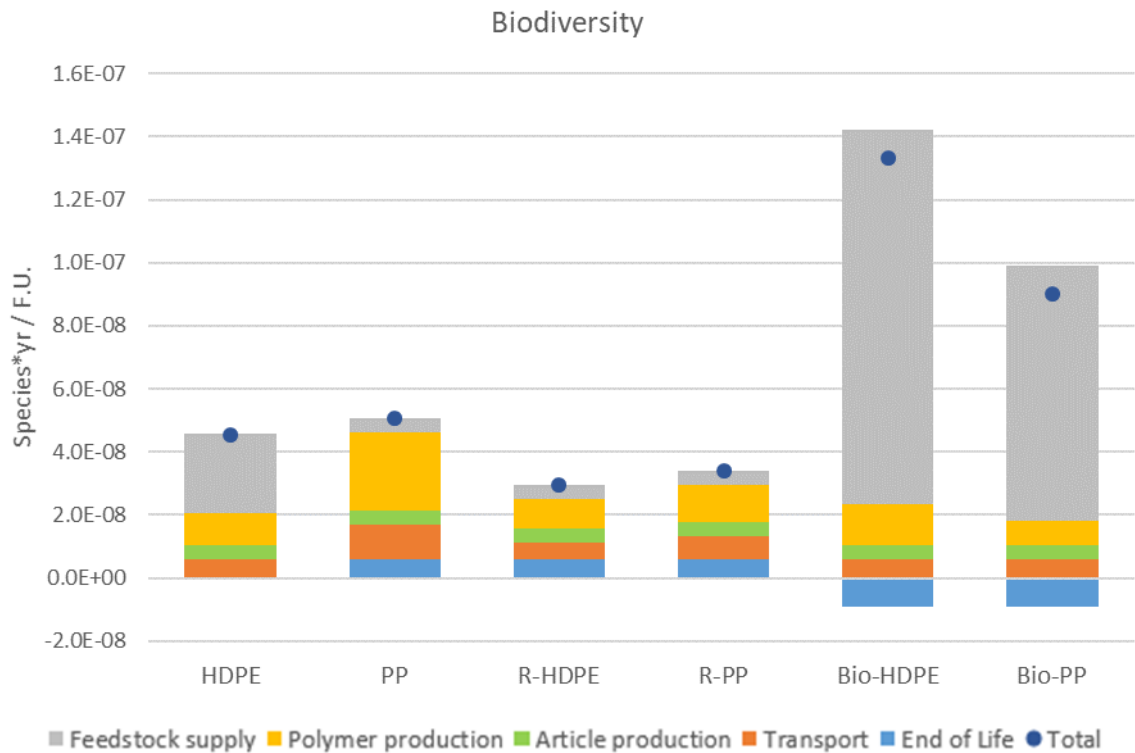
**Table 12.8.** iLUC contribution to the potential Climate Change impact of monobloc stacking chairs LCA scenarios.

Scenario / Polymer	iLUC Climate Change impact [kg CO <sub>2</sub> eq/FU]	Total Climate Change impact (incl. iLUC) <sup>(1)</sup> [kg CO <sub>2</sub> eq/FU]
S1 – HDPE	-	(7.85)
S2 – PP	-	(8.14)
S3 – R-HDPE	-	(5.47)
S4 – R-PP	-	(6.20)
S5 – Bio-HDPE	1.46	18.5 (17.0)
S6 – Bio-PP	1.25	12.65 (11.4)

<sup>(1)</sup> Values in parenthesis refer to the Climate Change impact without the iLUC contribution.

### 12.6.2 Biodiversity

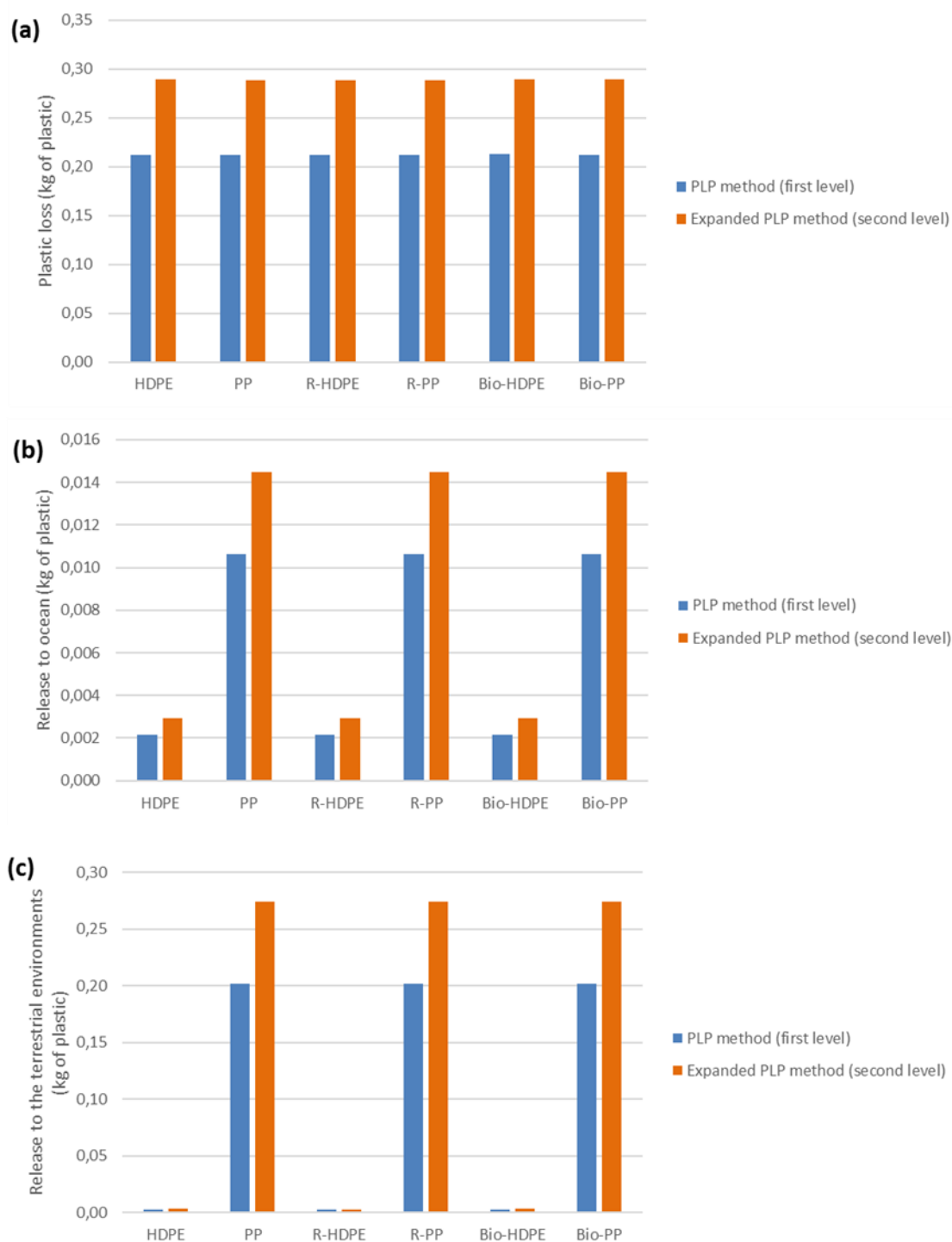
Estimated potential Biodiversity impacts of the investigated scenarios are presented in Figure 12.14. It is reminded that the assessment is performed through an endpoint-level indicator relying on impact assessment methods that for some of the underlying midpoint impact drivers differ from those recommended in the present method.



**Figure 12.14.** Potential biodiversity impact of monobloc stacking chairs LCA scenarios, expressed as potential loss of species per year.

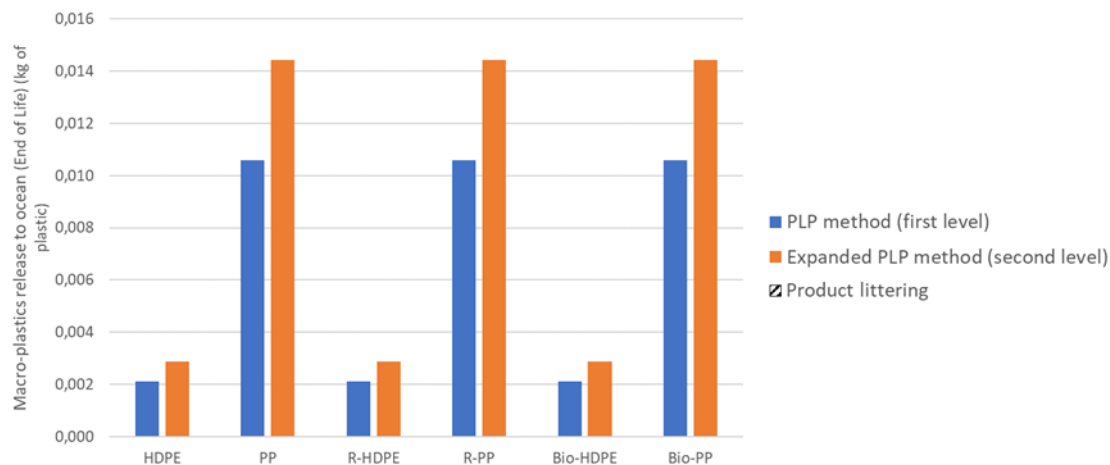
**12.6.3 Macro- and micro-plastics generation**

Figure 12.15 shows the total plastic loss, release to ocean and release to the terrestrial environment estimated for the assessed chairs scenarios, considering the whole product life cycle. Estimates obtained by applying all the two different explored approaches are presented, including the total loss and release to ocean/terrestrial environment calculated through both the PLP method (first level) and the Expanded PLP method (second level).



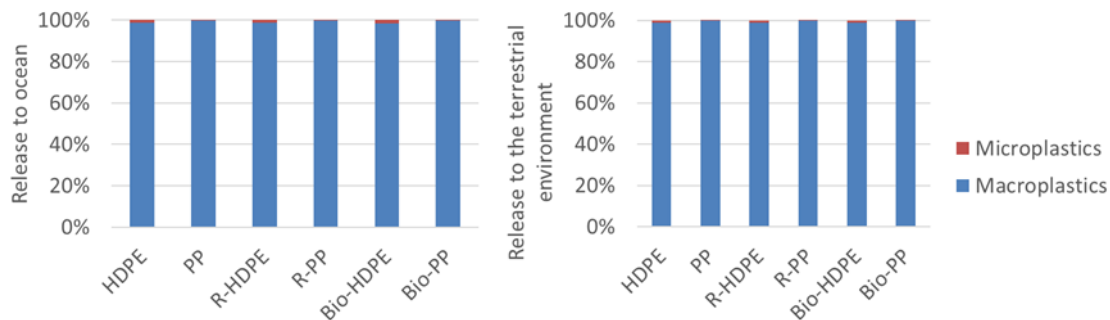
**Figure 12.15.** Total plastic loss (a), release to ocean (b) and release to the terrestrial environment (c) estimated for chairs LCA scenarios with different approaches. The contribution of both macro- and micro-plastics is included, considering the whole product life cycle.

Regarding the sole release of macro-plastics to ocean at End of Life (from product littering and waste mismanagement), Figure 12.16 presents the contribution of each chairs scenario, estimated with the two different explored approaches: the PLP method (first level), the Expanded PLP method (second level).



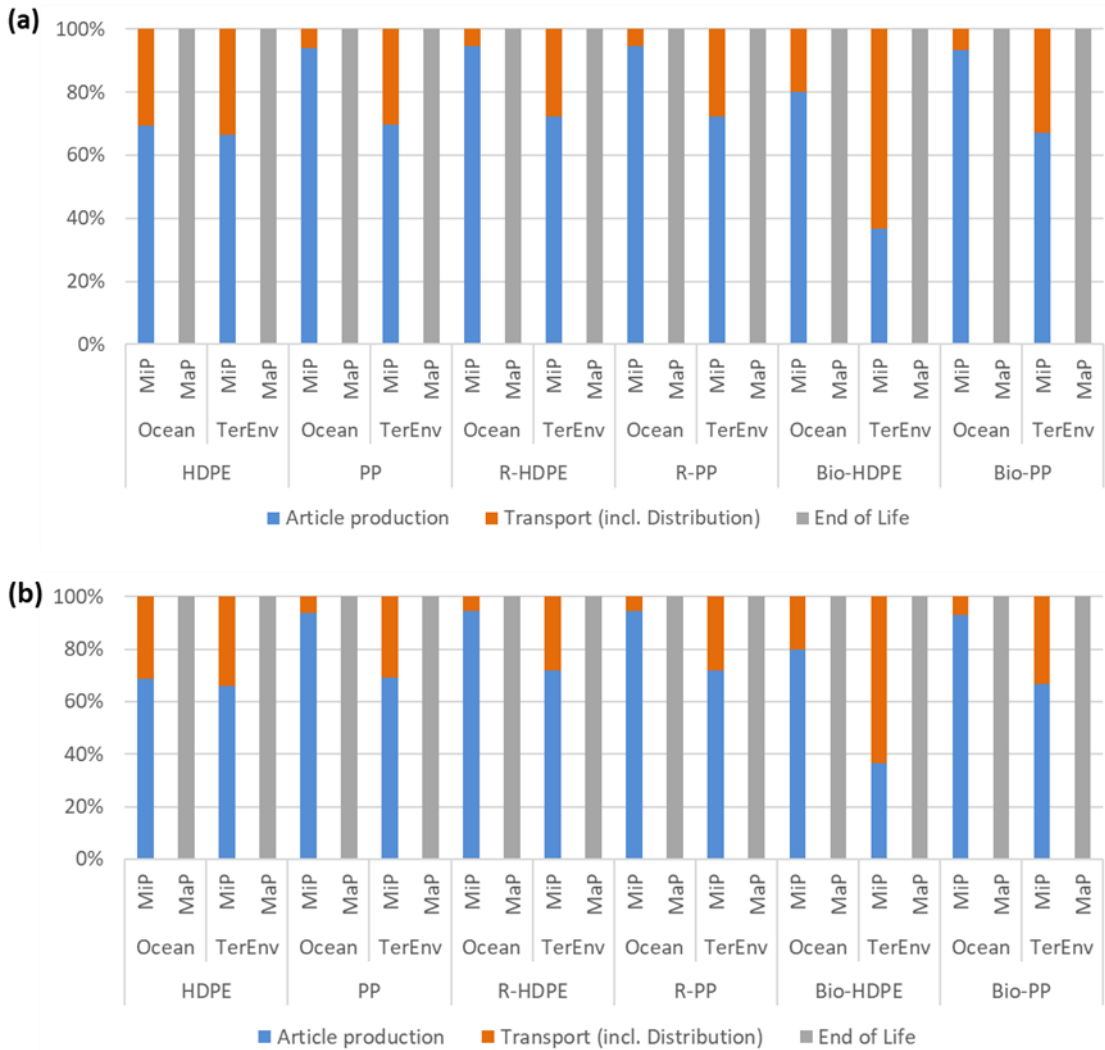
**Figure 12.16.** Macro-plastics release to ocean at End of Life, estimated with different approaches.

As for the share between the release of macro- and micro-plastics, Figure 12.17 shows as an example the relative contribution of the two categories to the total release to ocean and to the terrestrial environment estimated through the PLP method (first level). Note that the results are similar to those obtained when applying the Expanded PLP method (second level), as further discussed in Section 12.7.7.



**Figure 12.17.** Contribution of macro- and micro-plastics to the total plastic release to ocean and to the terrestrial environment estimated with the PLP method for each chairs LCA scenario.

The contribution of the different lifecycle stages to the total macro- and micro-plastics release to ocean and to the terrestrial environment of the assessed chairs scenarios has been finally evaluated. The results of this “hotspot analysis” are reported in Figure 12.18 for both the PLP method and the Expanded PLP method.



**Figure 12.18.** Contribution of life cycle stages to the total macro-plastics (MaP) and micro-plastics (MiP) release to ocean (Ocean) and to the terrestrial environment (TerEnv) estimated for chairs LCA scenarios with (a) the PLP method (first level) and (b) the Expanded PLP method (second level).

## 12.7 Interpretation

In the interpretation of case study results, most relevant impact categories and lifecycle stages of the studied systems are firstly identified (12.7.1 and 12.7.2, respectively). Scenario impacts are then compared, considering both characterised LCIA results (12.7.3) and total normalised and weighted impacts (12.7.4). Results obtained by applying individually each viable End of Life option are separately discussed in Section 12.7.6, while the effects of iLUC and the contribution to macro- and micro-plastics generation are addressed in Sections 12.7.5 and 12.7.7, respectively.

Most relevant processes were not identified, since it was not possible to achieve a similar level of vertical disaggregation for foreground processes included in the life cycle models of the analysed scenarios (e.g. a higher disaggregation could be achieved for bio-based HDPE chairs, but this was not the case of the other scenarios). Therefore, the identification of most relevant processes cannot be carried out consistently across all scenarios, and a more detailed investigation for specific scenarios would not be meaningful. The identification of most relevant elementary flows was also not undertaken, as this would require prior identification of most relevant processes. Note, however, that any company, organisation or any other supply chain actor applying the



present method shall proceed with the identification of both most relevant processes and elementary flows.

### 12.7.1 Identification of most relevant impact categories

Table 12.9 shows the most relevant impact categories identified for each chairs scenario, based on normalised and weighted impacts, according to the approach outlined in the Method (Report I, Section 6.2.1). Relevant categories were identified as those that cumulatively contribute to at least 80% of the total normalised and weighted impact of the considered impact categories, excluding toxicity-related ones (these being still based on characterisation factors implemented in the EF 2.0 impact assessment methods, and hence not yet updated based on REACH data<sup>218</sup>). 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 except the bio-based ones, i.e. Bio-HDPE and Bio-PP chairs. In those cases the most relevant category is by far (65% and 69% of the total impact, respectively) Particulate Matter, followed by Climate Change.

**Table 12.9.** Most relevant impact categories identified for monobloc stacking chairs LCA scenarios and related contribution to the total normalised and weighted scenario impact.

S1 - Fossil-based HDPE		S2 – Fossil-based PP		S3 - Recycled HDPE	
<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%	Climate Change	41%	Climate Change	53%
Climate Change	40%	Resource Use - fossils	41%	Resource Use - fossils	27%
Particulate Matter	4%	Acidification	4%	Particulate Matter	4%
<b>Total</b>	<b>85%</b>	<b>Total</b>	<b>86%</b>	<b>Total</b>	<b>84%</b>
S4 – Recycled PP		S5 – Bio-based HDPE		S6 – Bio-based PP	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Climate Change	50%	Particulate Matter	65%	Particulate Matter	69%
Resource Use - fossils	30%	Climate Change	14%	Climate Change	12%
Particulate Matter	4%	Eutrophication - marine	4%	Eutrophication - marine	4%
<b>Total</b>	<b>84%</b>	<b>Total</b>	<b>82%</b>	<b>Total</b>	<b>85%</b>

### 12.7.2 Identification of most relevant life-cycle stages

Table 12.10 shows the most relevant lifecycle stages in the relevant impact categories identified in Section 12.7.1, and the associated contribution, quantified according to the rules detailed in the Method (Report I, Section 6.2.2). Most relevant stages include those that together contribute to at least 80% of the total lifecycle impact in the specific

<sup>218</sup> According to the latest version of the Product Environmental Footprint Category Rules Guidance (EC, 2018), toxicity-related impact indicators based on these characterisation factors shall be excluded from the procedure to identify most relevant impact categories

category, and are highlighted in yellow in Table 12.10. For a more exhaustive picture, the contribution of all lifecycle stages is reported.

For scenarios relying on virgin fossil-based polymers (i.e. HDPE and PP chairs), the two most relevant life cycle stages identified for most relevant impact categories are Polymer Production and Feedstock Supply, with an order depending on the category. More specifically, in the case of Climate Change, Polymer Production is the most relevant stage, contributing to 36-39% of the total impact score, followed by Feedstock Supply (19-20%), as well as by Distribution (18-19%) and Article Production (15-16%). Concerning Resource Use - fossils, Feedstock Supply is identified as the most relevant life cycle stage, dominating the total impact with a 77% contribution in case of HDPE, and with 78% contribution in case of PP.

For scenarios relying on recycled polymers (i.e. R-HDPE and R-PP chairs), Distribution is instead identified, along with Article or Polymer Production, as one of the two most relevant stages across the majority of most relevant categories. Exception is Resource Use – fossils, where Feedstock Supply still dominates the total impact.

The picture is different in the case of Bio-HDPE and Bio-PP chairs, where Feedstock Supply is by far the most relevant lifecycle stage, contributing 89-100% of the total impact score in all the relevant impact categories (i.e. Particulate Matter, Climate Change and Eutrophication - marine). This finding can be attributed, on the one hand, to the relatively high feedstock consumption for the production of the two bio-based polymers and on the other one to the emissions from the pre-harvest burning practice applied to 45% of Brazilian sugarcane.

**Table 12.10.** Contribution of life cycle stages to the total characterised impacts of the most relevant categories identified for monobloc stacking chairs LCA scenarios. Most relevant stages (i.e. those contributing to more than 80% of the total impact) are highlighted in yellow.

S1 – Fossil-based HDPE					
Resource Use - fossils		Climate Change		Particulate Matter	
Life cycle stage	Contrib.	Life cycle stage	Contrib.	Life cycle stage	Contrib.
Feedstock Supply	77.4%	Polymer Production	35.9%	Polymer Production	50.0%
Polymer Production	23.6%	Feedstock Supply	19.8%	Feedstock Supply	24.9%
Article Production	11.8%	Distribution	18.7%	Distribution	24.4%
Distribution	11.4%	Article Production	15.7%	Article Production	24.0%
End of Life	-24.3%	End of Life	9.9%	End of Life	-23.2%
S2 – Fossil-based PP					
Climate Change		Resource Use - fossils		Acidification	
Life cycle stage	Contrib.	Life cycle stage	Contrib.	Life cycle stage	Contrib.
Polymer Production	38.7%	Feedstock Supply	75.7%	Polymer Production	37.4%
Feedstock Supply	18.8%	Polymer Production	25.5%	Feedstock Supply	33.8%
Distribution	18.1%	Article Production	11.9%	Distribution	28.1%
Article Production	15.2%	Distribution	11.3%	Article Production	20.9%
End of Life	9.2%	End of Life	-24.4%	End of Life	-20.2%

<b>S3 – Recycled HDPE</b>					
<b>Climate Change</b>		<b>Resource Use - fossils</b>		<b>Particulate Matter</b>	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Distribution	26.9%	Feedstock Supply	77.6%	Distribution	45.9%
Polymer Production	26.0%	Article Production	33.5%	Article Production	45.1%
Article Production	22.5%	Distribution	32.3%	Polymer Production	33.6%
End of Life	14.2%	Polymer Production	25.3%	Feedstock Supply	19.0%
Feedstock Supply	10.4%	End of Life	-68.7%	End of Life	-43.7%
<b>S4 – Recycled PP</b>					
<b>Climate Change</b>		<b>Resource Use - fossils</b>		<b>Particulate Matter</b>	
<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	32.7%	Feedstock Supply	74.3%	Distribution	40.4%
Distribution	23.7%	Polymer Production	28.6%	Article Production	39.9%
Article Production	19.8%	Article Production	25.3%	Polymer Production	30.8%
End of Life	12.1%	Distribution	24.5%	Feedstock Supply	21.2%
Feedstock Supply	11.6%	End of Life	-52.7%	End of Life	-32.5%
<b>S5 – Bio-based HDPE</b>					
<b>Particulate Matter</b>		<b>Climate Change</b>		<b>Eutrophication - marine</b>	
<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	99.6%	Feedstock supply	88.7%	Feedstock Supply	98.1%
Distribution	0.3%	Distribution	8.6%	Distribution	1.5%
Article Production	0.3%	Article Production	7.2%	Article Production	0.6%
Polymer Production	0.2%	Polymer Production	6.2%	Polymer Production	0.6%
End of Life	-0.3%	End of Life	-10.7%	End of Life	-0.8%
<b>S6 – Bio-based PP</b>					
<b>Particulate Matter</b>		<b>Climate Change</b>		<b>Eutrophication - marine</b>	
<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	97.6%	Feedstock Supply	97.4%	Feedstock Supply	90.1%
Polymer Production	2.1%	Distribution	12.9%	Polymer Production	8.4%
Distribution	0.3%	Article Production	10.8%	Distribution	1.7%
Article Production	0.3%	Polymer Production	-4.7%	Article Production	0.7%
End of Life	-0.4%	End of Life	-16.4%	End of Life	-0.9%

1

## 2 12.7.3 Interpretation of characterised results

3 In this section, the characterised potential impacts of the assessed scenarios (Figures  
4 12.7-12.9 and Table B.9.7) are compared to evaluate the effects of using alternative  
5 feedstock sources or materials for a specific fossil-based polymer of reference used in  
6 chair manufacturing. Therefore, the impacts of recycled and bio-based HDPE chairs are  
7 compared with those of virgin fossil-based HDPE chairs, and the same is done for PP-

based chairs from different feedstock sources. The impacts of reference scenarios (fossil-based HDPE and PP chairs) are also initially compared, to provide useful elements to potentially extend the comparative considerations provided below to any of such reference scenarios.

The comparison focuses on the sixteen, default impact categories considered in the Product Environmental Footprint (PEF) context and in this assessment. Potential impacts on Biodiversity are discussed separately, as being estimated via an endpoint indicator that is not recommended in the PEF framework, and partially relying on different impact assessment methods for the underlying midpoint-level impact categories. In the following comparison, differences between scenario impacts lower than 10% were not considered significant, in light of the uncertainty associated with the developed life cycle models and the applied impact assessment models. Moreover, any comparative considerations related to Human Toxicity and Ecotoxicity impact categories need to be interpreted in light of the higher uncertainty of the underlying impact assessment models and results. The same applies, to a lower extent, also to Water Use and Land Use.

Focusing on the relative performance of the two fossil based-reference scenarios, no significant differences are observed between HDPE and PP chairs in the vast majority of the assessed impact categories. The only exceptions are Ozone Depletion, Particulate Matter and Photochemical Ozone Formations, where HDPE chairs show a moderately better performance (12-18% impact reduction).

The use of 100% recycled material in HDPE or PP chair manufacturing provides more or less important environmental benefits compared to the use of virgin material, if we exclude Ozone Depletion (where a 70% impact increase is observed) and Land Use (where no relevant changes occur). The worsened Ozone Depletion performance can be attributed to the increased Polymer Production impacts, which are dominated by the life cycle of Sodium Hydroxide used in mechanical recycling of both HDPE and PP. Leaving the discussed exceptions apart, an impact reduction ranging from 18% to 65% is observed for HDPE chairs when recycled material is used as an input. In the case of PP chairs, a similar decrease is also achieved (15-54%).

Replacing fossil-based HDPE with bio-based HDPE as a chair material is proven to be beneficial only in the category of Resource Use - fossils, thanks to the full conversion to a bio-based feedstock source. Conversely, a substantially worsened performance is observed in the remaining categories, due mainly to the increased impact from Feedstock Supply and/or, in some categories, of Polymer Production. The increase is particularly huge in a number of categories more largely affected by agricultural production, including Land Use, Particulate Matter (dominated by emissions from pre-harvest sugarcane burning), marine and freshwater Eutrophication (nutrient emissions) and Human Toxicity – non-cancer (metal emissions). However, a relevant increase is also observed in Eutrophication – terrestrial (nutrient emissions), Photochemical Ozone Formation (CO and NOx emissions), and Acidification. For the remaining categories, the increase is lower, but still significant, ranging between 27% (Ionising Radiation) to 284% (Ozone Depletion). A similar picture of results is observed also for Bio-PP chairs, although the relative impact increase achieved in this case compared to the use of fossil-based PP is moderately lower (to an extent depending on the category).

Focusing briefly on estimated potential impacts on biodiversity, both recycled HDPE and PP chairs show a moderately reduced impact (-40% approximately) compared to their virgin, fossil-based counterpart. This reflects the lower burdens from Feedstock Supply and Polymer Production (which are less resource- and emission-intensive), rather than actual differences in supply chain drivers. Conversely, bio-based HDPE and PP chairs involve a higher biodiversity impact (especially when Bio-HDPE is used), due especially to the increased impact from Feedstock Supply, and the resulting land occupation for sugarcane growing. It has to be reminded, however, that direct potential biodiversity (ecosystem) impacts from oil leakage are not quantified for chairs relying on fossil-based polymers considered as a reference (although emissions from leakage per unit of oil supplied are reported to be quite small; see Section 4.4.1.1).

As a last comment, it is noted how the Climate Change impact of bio-based chairs (i.e. Bio-HDPE and Bio-PP chairs) is moderately reduced if the contribution of the missed release of biogenic carbon after 100 years from landfilling is taken into account. Indeed, a decrease between 13% (Bio-HDPE) and 18% (Bio-PP) is observed. This result should be interpreted keeping in mind the relative high share of landfilling assumed in the modelled EU-average scenario (31%), which should be progressively reduced in the near future.

#### **12.7.4 Interpretation of normalised and weighted results**

This section briefly compares the investigated scenarios based on the respective total impact scores, calculated by aggregating normalised and weighted impact assessment results across all impact categories (Table B.9.9 in Annex B.9). The comparison is carried out according to the same criteria outlined in Section 12.7.3 in terms of contrasted chair scenarios, and considering significant only impact differences larger than 10%. While affected by greater uncertainty (from the calculation and application of normalisation and weighting factors), total impact scores provides a more immediate and synthetic indication of the overall environmental performance of the analysed product scenarios. This facilitates the interpretation compared to looking at characterised results from several impact categories. However, such estimates are inevitably affected by value choices underlying the definition of weighting factors aimed at establishing an order of relevance for the different impact categories in a European decision context.

The comparison among total normalised and weighted scenario impacts reflects the overall picture of results emerged when focusing on characterised impact assessment results in single impact categories, and can be summarised in the following points:

- The two fossil-based scenarios, namely HDPE and PP chairs, show a comparable overall impact.
- The impact of chairs entirely relying on recycled polymers (i.e. R-HDPE and R-PP) is considerably lower than their virgin counterparts (i.e. almost half compared to that of virgin HDPE and PP chairs).
- The bio-based scenarios (i.e. Bio-HDPE and Bio-PP chairs) show a substantially worse performance than their fossil-based counterparts. Specifically, Bio-HDPE chairs presents a six times higher impact score than HDPE chairs (showing the worst absolute performance) whereas the total impact of Bio-PP chairs is five times higher than that of PP chairs.

#### **12.7.5 Effects of indirect Land Use Change (iLUC)**

When the contribution of GHG emissions from iLUC is taken into account, only a small changes are identified in the Climate Change impact of the bio-based alternatives (Figure 12.7 and Table 12.8). An increase of 8% and 11% is indeed observed for Bio-HDPE and Bio-PP chairs, respectively. Overall, accounting for iLUC effects does not affect the relative performance of the assessed scenarios (i.e. their comparison is not affected). However, this outcome needs to be read also in light of the GHG emission factors applied for iLUC in this assessment, which appear to fall in the lower end of the range of values available in the recent literature (see Section 4.4.16.3 of the Method (Report I)).

#### **12.7.6 Alternative End of Life Options**

This section discusses characterised scenario results calculated by individually applying each viable End of Life option as an exclusive (100%) End of Life scenario (as presented in Figures 12.11 to 12.13). The main purpose is to evaluate how the performances of single chairs scenarios are affected by the change in the applied End of Life option, rather than to mutually compare such scenarios for different End of Life options. However, the results should not be interpreted as a strict comparison among different End of Life alternatives for the studied article, since the evaluation applies a product perspective,

and burdens/benefits of selected End of Life options (e.g. recycling) are partitioned between different product life cycles (which prevents from capturing the full implications of having a given waste stream routed to a specific End of Life option). In a waste management system perspective, where allocation is not needed, the total (system-wise) savings associated with the End of Life pathway “100% recycling” would likely be higher than what presented in this report where a product-perspective is applied.

Out of the three alternative product End of Life scenarios (i.e. 100% mechanical recycling, 100% incineration and 100% landfilling), none can be identified as preferable across all the assessed impact categories and chairs scenarios. For all of these, the 100% recycling scenario is preferable in a few categories, including Human Toxicity – cancer, Ecotoxicity – freshwater, Water Use and Resource Use – fossils. For scenarios relying on virgin or recycled fossil-based polymers, this is also the case of Climate Change, Human Toxicity – non-cancer and Eutrophication – freshwater. On the other hand, recycling is the worst scenario in terms of Ozone Depletion and, limited to bio-based HDPE and PP chairs, Land Use. For the remaining categories or scenarios, recycling shows an intermediate performance between incineration and landfilling.

The incineration scenario is preferable in a considerable number of impact categories (i.e. nine out of sixteen)<sup>219</sup> for all the assessed scenarios, while for bio-based chairs the same applies also to Human Toxicity – non-cancer and Eutrophication – freshwater. However, incineration shows the worst performance in terms of Water Use and, limited to scenarios relying on virgin and recycled fossil-based polymers, also of Climate Change.

In most impact categories, 100% landfilling is the least favourable End of Life scenario, in line with the, priority order outlined in the waste hierarchy, which set disposal as the least preferable option (EC, 2008). The only exceptions are Ozone Depletion, Water Use and Climate Change (if bio-based scenarios are excluded), where other End of Life scenarios show the worst performance. Moreover, when the contribution of non-released biogenic carbon after 100 years from landfilling is taken into account, landfilling of bio-based HDPE and bio-based PP chairs is preferable in terms of Climate Change.

### **12.7.7 Interpretation of results related to macro- and micro-plastics generation**

This section discusses the results presented in Section 12.6.3 on the estimated potential generation of macro- and micro-plastics of the assessed chairs scenarios. The latter are compared according to the same criteria applied to the interpretation of the potential environmental impacts (Section 12.7.3), initially focusing on the results from the two-level approach applied as a base case. No sensitivity analyses were assessed for this case study.

Regarding the total loss of plastics (macro- and micro-plastics) (Figure 12.15), the two base-case approaches (PLP method and Expanded PLP method) provide similar results when comparing the different chairs scenarios. In both cases, no differences are observed among the different scenarios (<0.2%). This is mostly associated to the fact that all scenarios require the same mass to fulfil the functional unit which contribute to the same loss of macro-plastics at the End of Life and limited differences are observed in the supply-chain of the scenarios leading to differences in the loss of micro-plastics related to transportation (e.g. tire abrasion). Note, however, that the applied methods rely on mass-based indicators (which are affected by differences in chair mass) rather than quantifying the number of items (or plastic fragments/particles) lost or released to the environment. If this was the case, all scenarios would result in the same macro-plastics generation at End of Life when applying both PLP method and Expanded PLP method, since chairs with a lower mass would not imply a reduction in the contribution of product littering and waste mismanagement (which are based on rates that are not

<sup>219</sup> These include, Ozone Depletion, Particulate Matter, Ionising Radiation, Photochemical Ozone Formation, Acidification, Eutrophication – terrestrial, Eutrophication – marine, Land Use and Resource Use – minerals and metals.

affected by the type of feedstock or material). In turn, an identical macro-plastics generation at End of Life would imply having a comparable total loss, these being mostly determined by macro-plastics (see below).

Conversely, the use of HDPE-based polymers (HDPE, R-HDPE and Bio-HDPE) result on the lowest plastic release to the ocean (-80% compared to PP-based scenarios) and to the terrestrial environment (-99%). This is related to the modelling parameters considered for the release rates in the PLP method, which are based on the economic value of the material and its recovery as resource input to the technosphere. HDPE is considered a material with high value leading to a release rate of 1% for both environmental compartments. This is a very low rate compared to the PP-based scenarios, which assume a release rate to the ocean of 5% and a release rate to the terrestrial environment of 95%. This highlights the high sensitivity of the results to the defined parameters.

Comparing the two base-case approaches, the Expanded PLP method results in a 36% higher plastic loss and release to ocean and the terrestrial environment compared to the original PLP method (Figure 12.15). This is mainly due to an increased contribution of macro-plastics loss at End of Life, which also accounts for mismanagement of plastic waste exported for (sub-standard) recycling outside the EU (where the overall waste mismanagement share is increased by 5-6% compared to the assumption that all plastic waste is recycled in Europe). Conversely, the two additional micro-plastics sources considered in this alternative approach (road markings and marine coatings) does not significantly affect the results, due to the overall modest contribution of micro-plastics to the total release (in terms of mass).

Focusing on the relation between total loss and release (which depends on the parameters of the PLP method), this depends on the polymer. Regarding PP, the plastic loss is modelled as fully released to the environment due to the lack of informal waste collection for this type of product. In relative terms, release to ocean accounts for a marginal share of the plastic loss (5%), while release to the terrestrial environment is the most relevant (95%). Regarding HDPE, the higher economic value leads to release rates of 1%. As shown in Figure 12.17 for the PLP method, the total plastic release to both ocean and the terrestrial environment is dominated by the release of macro-plastics at the End of Life (more than 99%), while micro-plastics released throughout the upstream life cycle have only a minor role. Similar results are also observed for the Expanded PLP method (not shown), despite two additional sources of micro-plastics are considered in the latter (i.e. road markings and marine coatings). In both cases, this is a consequence of the prevailing mass of macro-plastics, which is directly related to the mass of chairs required per functional unit (reference flow), by means of higher loss and release rates compared to micro-plastics. Therefore, the mass of macro-plastics is at least one order of magnitude higher than the mass of micro-plastics, which instead depends on the quantity of relevant lifecycle processes by means of (much) lower loss and release coefficients (depending on the source).

With regard to the origin of released plastics (in terms of life cycle stage and source), macro-plastics entirely originate from the End of Life stage (Figure 12.18), as they are only derived from mismanagement of the product as waste. Note that no littering is assumed for chairs as it is considered that chairs are managed as part of cars at the End of Life thereby following specific collection and management pathways. On micro-plastics release, the relative contribution of the different sources (e.g. loss of plastic pellets and tire abrasion) depends on the characteristics of the supply-chain (e.g. feedstock origin, which affect the contribution from transport across the life cycle), as well as on the mass of polymer required per functional unit (which affect, to different extents, all the considered sources). However, within the limited role of micro-plastics in the overall release to the environment, the most relevant contribution is generally provided by pellet losses from the Polymer and Article Production stages, while transport-related sources have a more restricted role. Exceptionally, transport emissions are relevant

1 (approximately 60%) for the use of Bio-HDPE scenario, whose supply-chain represents  
2 larger transport distances.

3 In conclusion, due to the dominant role of macro-plastics loss and defined parameters for  
4 release rates to the environment at the End of Life of chairs, a best overall performance  
5 is associated to those scenario employing polymers with a higher economic value and  
6 with the lowest release rates. In this context, the use of fossil-based, recycled and bio-  
7 based HDPE chairs are the scenarios with the lowest plastic release to both ocean and  
8 the terrestrial environment, although all scenarios yield similar plastic loss results. From  
9 a methodological perspective, the different approaches provide similar results when  
10 comparing alternative scenarios.



## 13 Case study 10: Wipes

This case study focuses on non-woven wipes for personal care, considered as a representative plastic article of the consumer goods sector. While wipes consisting of different types of materials are available on the market (polymer-based, textiles, a mixture of them etc.), this study considers wipes manufactured exclusively by a single, polymer-based material, consistently with the focus of the project on plastic articles, and to avoid unnecessarily increasing the complexity of the study.

Availability of data on the specific material composition of polymer-based wipes, on material quantities, and on the wipes manufacturing process is very scarce. Therefore, several assumptions and approximations had to be performed throughout the study, and the results presented in this section need to be interpreted in light of these limitations.

### 13.1 Assessed scenarios

The use of different materials and/or feedstock for wipes manufacturing was explored by assessing a number of alternative scenarios (Table 13.1). Two reference scenarios based on fossil-based plastics (PP and LDPE) were analysed first. The use of two fully bio-based, drop-in alternatives to PP and LDPE (Bio-PP and Bio-LDPE) is modelled in Scenarios 3 and 4, respectively. In this case, bio-Ethylene derived from Brazilian sugarcane is used as a feedstock.

**Table 13.1.** LCA scenarios assessed for the wipes case study.

Scenario	Polymer	Monomer(s)	Feedstock	End of Life options <sup>(1)</sup>
1 - Conventional polymer 1	PP	Propylene	Fossil-based (oil/natural gas)	Incineration Landfilling
2 - Conventional polymer 2	LDPE	Ethylene	Fossil-based (oil/natural gas)	Incineration Landfilling
3 - Alternative polymer 1	Bio-PP	Propylene	Sugarcane (Brazil)	Incineration Landfilling
4 - Alternative polymer 2	Bio-LDPE	Bio-ethylene	Sugarcane (Brazil)	Incineration Landfilling

<sup>(1)</sup> The impacts of scenarios were individually assessed for each listed End of Life option, as well as for a combination of such options reflecting as far as possible the average situation at the EU level.

### 13.2 Functional Unit and reference flow

Wipes are a personal care product that is mainly intended for skin cleaning purposes. The functional unit was thus defined as “*providing personal cleaning by means of 1000 single-use wipes with an average area of 350 cm<sup>2</sup> to people living in the EU, ensuring a proper cleaning and skin hydration*”. A more detailed description of the functional unit assumed for wipes is provided in Table 13.2.

1 **Table 13.2.** Definition of the functional unit for wipes LCA scenarios.

Aspect	Description
“What” (function(s) or service(s) provided)	Providing personal cleaning by means of non-woven, disposable wipes with a cleaning area of 350 cm <sup>2</sup>
“How much” (extent of the function(s) or service(s))	1000 wipes
“How well” (expected level of quality)	Providing proper cleaning and skin hydration to the user
“How long” (duration/lifetime of the function or service)	One time (single-use)
“Where” (location/geography of the service)	In the EU

2 The reference flow (i.e. the amount of wipes material required in order to fulfil the  
3 functional unit), was defined according to technical details found in the literature and in  
4 factsheets of non-biodegradable (fossil-based) wipes available on the market. In this  
5 respect, it is noted that it was not possible to calculate the mass of wipes material in  
6 such a way that they provide a comparable cleaning performance. The same mass was  
7 this assumed for all the assessed materials (PP, LDPE, Bio-PP and Bio-LDPE). Table 13.3  
8 summarises the reference flow of each scenario.

9 **Table 13.3.** Calculation of the reference flow for wipes LCA scenarios.

Polymer	Wipes mass (kg)	Reference flow (kg/FU)
PP	0.00245	2.45
LDPE	0.00245	2.45
Bio-PP	0.00245	2.45
Bio-LDPE	0.00245	2.45

10

### 11 **13.3 System boundary**

12 In all scenarios, the system boundary was set in order to cover the most relevant stages  
13 and processes of the full product life cycle (cradle-to-grave perspective), as described  
14 below and depicted in Figures 13.1 to 13.4.

- 15 1. *Feedstock Supply*<sup>220</sup> – covering extraction, transport and possible refining of  
16 crude oil and natural gas (fossil-based polymers), crop cultivation (bio-based  
17 polymers), as well as transport of these feedstock sources to downstream  
18 conversion processes (e.g. naphtha cracking, sugarcane fermentation);
- 19 2. *Polymer Production*<sup>221</sup> – covering all the activities associated with the  
20 conversion of feedstock materials into the relevant monomer(s) and final

<sup>220</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

<sup>221</sup> Corresponding to the default stage of “Raw Material Acquisition and Pre-Processing” specified in the Method and in the PEF framework.

polymer, including any transport among these activities and final transport of  
polymer granulate to downstream manufacturing processes;

3. *Article Production*<sup>222</sup> – including wipes manufacturing through film extrusion  
(blowing) of polymer granulates;

4. *Distribution* – including transport of wipes from the manufacturing site to  
retailers and from these to final consumers;

5. *End of Life* – covering collection, transport, incineration, or disposal of wipes  
after use, including any avoided processes from energy substitution.

A different nomenclature was applied for some life cycle stages compared to the default  
nomenclature specified in the Method (and in the PEF framework) to make it more  
relevant for the investigated supply chains and the project scope. Moreover, the default  
“Raw Material Acquisition and Pre-processing” stage was further split into two separate  
sub-stages (i.e. Feedstock Supply and Polymer Production), to allow disaggregating the  
impacts of feedstock supply from downstream conversion processes, and hence to better  
appreciate any differences among the use of different feedstock sources (in line with the  
project goal).

The Use stage was excluded in this LCA study, as it can be reasonably assumed that the  
different wipes materials investigated in this study does not affect activities related to the  
Use stage. Therefore, these activities can be excluded from a comparative assessment,  
as being identical for all the examined product systems.

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 wipes,  
of the examined polymers and of plastics in general, as well as on their release and fate  
over the product life cycle. This is acknowledged as a limitation of this study, as additive  
production can account for a non-negligible portion of cradle-to-gate Climate Change  
impact and energy demand, which is up to 46% for (starch-based) polymer grades  
including larger shares of additives (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 plastics routed to biological treatments or subject to in-situ  
degradation).

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<sup>222</sup> Corresponding to the default stage of “Manufacturing” specified in the Method and in the PEF framework.

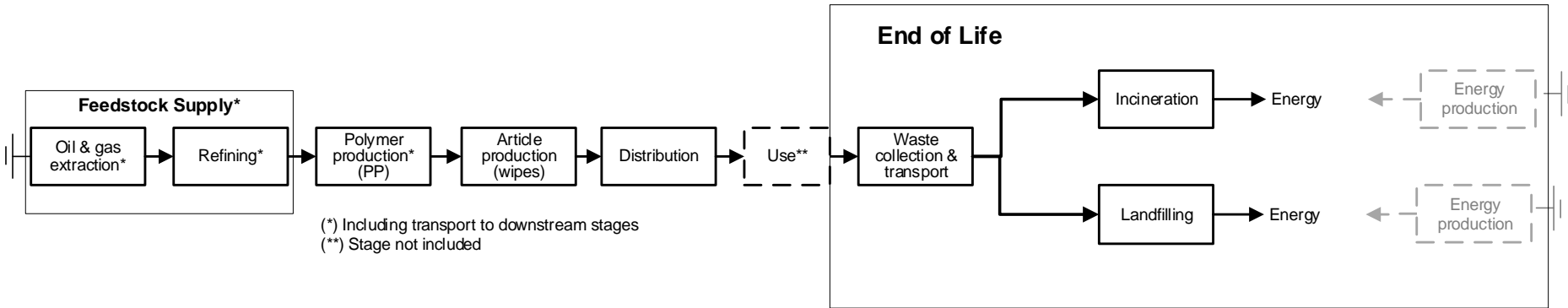


Figure 13.1. System boundary for fossil-based PP wipes (Scenario 1).

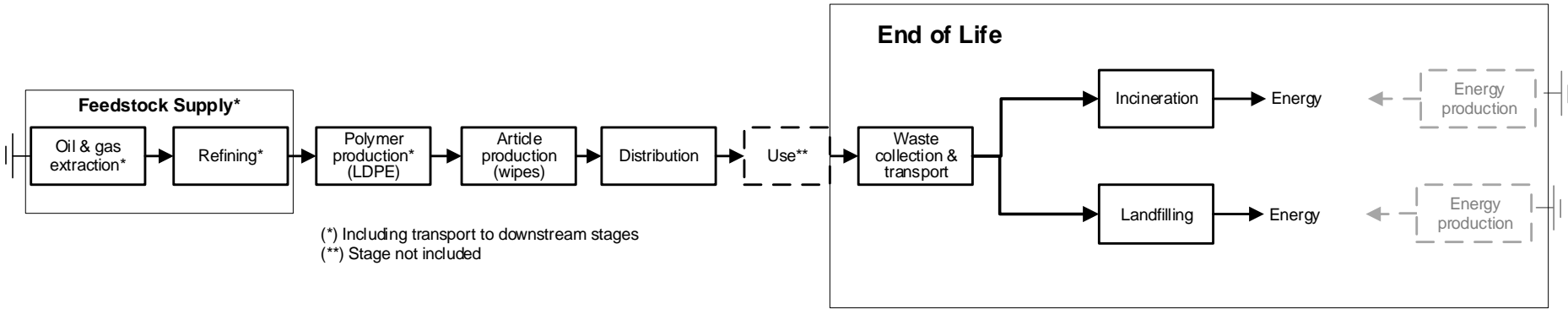


Figure 13.2. System boundary for fossil-based LDPE wipes (Scenario 2).

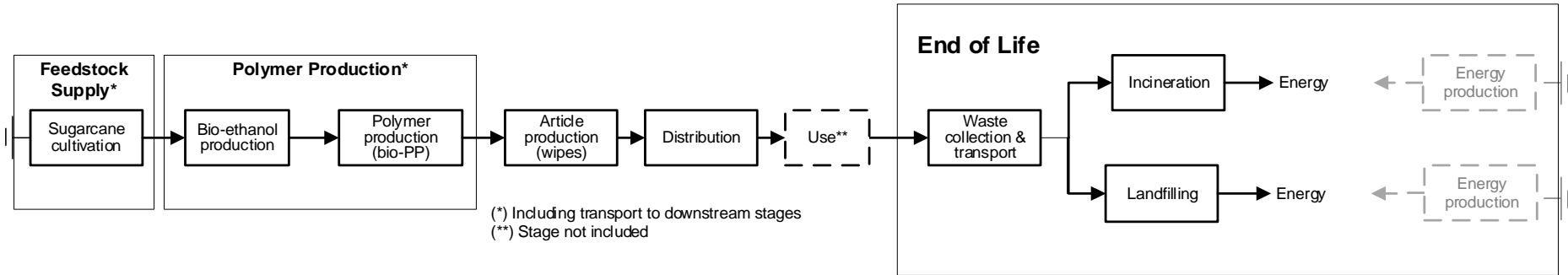


Figure 13.3. System boundary for bio-based PP wipes (Scenario 3).

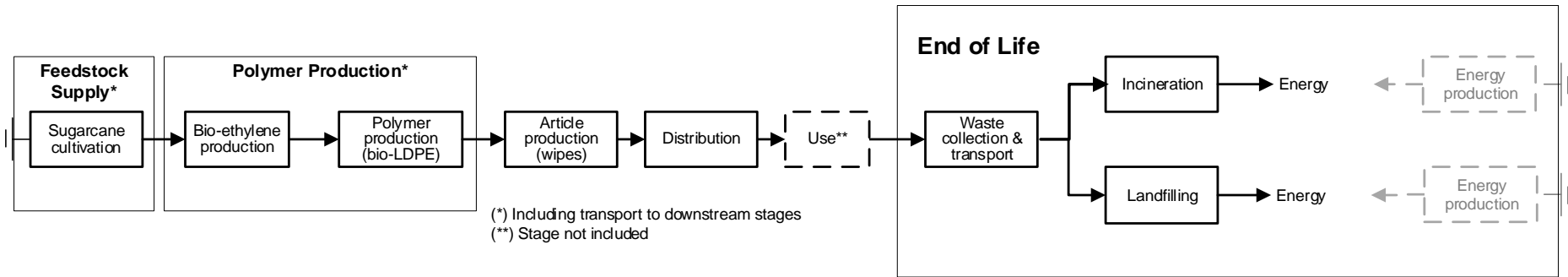


Figure 13.4. System boundary for bio-based LDPE wipes (Scenario 4).

## **13.4 Life Cycle Inventory**

This section describes the overall approach for developing the Life Cycle Inventory of the analysed scenarios, along with the related assumptions and data sources. The description is separated by major lifecycle stages, in the following sub-sections (13.4.1 – 13.4.7). The list of processes, related data sources, and main modelling details are provided in Annex B.10, Tables B.10.1 to B.10.4.

### **13.4.1 Feedstock Supply Stage**

#### **13.4.1.1 Fossil-based polymers**

For fossil-based polymers (PP and LDPE), the stage of Feedstock Supply includes the processes of crude-oil and natural gas extraction and transport, naphtha production in crude oil refineries, and its subsequent transport to downstream conversion processes (i.e. naphtha cracking). The modelling of these activities was carried out as described in Section 4.4.1.1, where the reader is referred to for further details on data sources, covered sub-processes and related environmental burdens, main modelling choices (e.g. allocation) and any adjustments performed in the implementation of the applied datasets.

#### **13.4.1.2 Bio-based polymers**

For the bio-based polymers (Bio-PP and Bio-LDPE), the stage of Feedstock Supply includes cultivation of the relevant crop and its subsequent transport to further processing in the same country.

Sugarcane cultivation was modelled through aggregated datasets from the GaBi database. Used in combination, these datasets depict a situation where 45% of sugarcane is manually harvested via the "slash and burn" practice, i.e. sugarcane residues (tops and leaves) are burned on standing plants before harvesting. This share was considered representative of the current situation. However, the slash and burn 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). Transport of harvested sugarcane to further processing in sugarcane mills for ethanol 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).

### **13.4.2 Polymer Production Stage**

The Polymer Production stage covers the activities of feedstock processing into relevant intermediates and monomer(s), the polymerisation process, as well as any transport among these activities and final transport of polymer granulate to the wipe manufacturing site. The following subsections (13.4.2.1 – 13.4.2.3) describe how these activities have been modelled in the present case study.

#### **13.4.2.1 Fossil-based polymers**

For conventional, fossil-based polymers (PP and LDPE), the whole process chain from feedstock processing to polymerisation, through the production of intermediates and monomers, was modelled by means of aggregated, gate-to-gate datasets provided by Thinkstep. Inputs include combinations of crude oil, natural gas and naphtha, depending on the polymer. All conversion processes are assumed to take place in Europe, so that the datasets reflect EU average background conditions in terms of e.g. energy generation, material supply and transport. For both polymers, the main conversion process involved in the supply chain is steam cracking of naphtha and natural gas, delivering the monomers ethylene, propylene, butadiene and other relevant intermediate flows (e.g. pyrolysis gas including mixtures of Benzene, Toluene and Xylenes). The

inventory of this process is mainly based on industry data, completed, where necessary, by literature data. 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 co-products. The final polymerisation process of PP and LDPE is mainly modelled based on relevant industry data from internationally adopted production processes, integrated with literature data when needed. No allocation is performed for this stage, being PP and LDPE the only outputs from the respective production process.

#### **13.4.2.2 Bio-based polymers**

##### **13.4.2.2.1 Bio-based PP**

The production of bio-based PP out of Brazilian sugarcane was modelled based on an aggregated dataset provided by Thinkstep. The dataset covers the steps of sugarcane processing to bioethanol in Brazil (fermentation and distillation), bioethanol transport to EU via transoceanic ship, conversion to bio-Ethylene (via dehydration) and subsequently to Propylene (via dimerization and metathesis), as well as final polymerisation to PP. 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. For the remaining conversion and polymerisation processes, no allocation or substitution needs to be applied. The inventory is mainly based on industry data from internationally adopted production processes, completed, where necessary, by literature data. The final (gas phase) polymerisation step specifically relies on secondary data. Background inventories from the GaBi database are used to model the burdens of the inventoried inputs and outputs.

##### **13.4.2.2.2 Bio-based LDPE**

The production of bio-based LDPE out of Brazilian sugarcane was modelled based on an aggregated dataset provided by Thinkstep. The dataset covers the steps of sugarcane processing to bioethanol in Brazil (fermentation and distillation), bioethanol transport to EU via transoceanic ship, conversion to bio-Ethylene (via dehydration), and its subsequent polymerisation to LDPE. 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. For the remaining conversion and polymerisation processes, no allocation nor substitution needs to be applied. The inventory is mainly 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.

##### **13.4.2.3 Transport of polymer granulate to the article production site**

Modelling of transport of polymer granulate from the polymerisation or compounding plant (inside or outside the EU), to the wipes manufacturing site in Europe, is based on the default transport scenario (distances and vehicle types) specified in the PEF context (and in the Method) for the route supplier-to-factory. In the case of polymers produced in Europe (i.e. all the polymers and copolymers investigated in this case study, except for PLA and the imported share of fossil-based PP and LDPE), the following routes were thus considered:

- i) 130 km by lorry (total weight >32 t; Euro 4);
- ii) 240 km by train (average freight); and
- iii) 270 km by ship (barge).

For the imported share of fossil-based PP and LDPE (10% and 14%, respectively), a transoceanic ship transport was considered as the main transport route. The corresponding overall 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). Countries contributing to at least 90% of the overall import were considered in the calculation, for an overall distance equal to 8,609 km for fossil-based PP, and to 6,528 km for fossil-based LDPE (see Tables 13.4 and 13.5). Oceanic ship transport was complemented with road transport to and from the harbour in both the exporting and importing country (i.e. the EU). Road transport is made by lorry (total weight >32 t; Euro 4) along an overall default distance of 1000 km.

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

**Table 13.4.** Calculation of the overall average sea distance for imports of fossil-based PP to Europe.

Exporting country	Import <sup>(1)</sup> (%)	Import (% cum.)	Distance <sup>(2)</sup> (km)	Weighted distance (km)
SAUDI ARABIA	41.3	41.3	8767.28	3622
KOREA, REPUBLIC OF (SOUTH KOREA)	13.5	54.8	16702.31	2259
INDIA	7.15	62.0	10267.28	734
RUSSIAN FEDERATION (RUSSIA)	6.75	68.7	5574.44	376
BRAZIL	6.63	75.4	10107.97	670
ISRAEL (GAZA and JERICOHO- >1994)	4.86	80.2	2987.48	145
SOUTH AFRICA (incl. NA - >1989)	4.26	84.5	12160.92	518
EGYPT	3.20	87.7	3212.14	103
UNITED STATES	2.99	90.7	6061.9	181
Other countries	9.32	100	-	-
<i>Overall weighted distance</i>				<i>8,609</i>

<sup>(1)</sup> Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

<sup>(2)</sup> 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 Middle-East and Asia were determined considering Marseille as destination port in Europe. For imports from other countries (in the case of PP Russia, Brazil, South Africa and United States), Rotterdam was considered as destination port.



**Table 13.5.** Calculation of the overall average sea distance for imports of fossil-based LDPE to Europe.

Exporting country	Import <sup>(1)</sup> (%)	Import (% cum.)	Distance <sup>(2)</sup> (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	-	-
Overall weighted distance				6,528

<sup>(1)</sup> Based on Comext data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). Reported shares were determined as 3-year averages of import shares calculated, based on raw Comext data, for the years from 2016 to 2018. Note that the total quantity of imports from extra-EU countries during a given year has been recalculated as the sum of imports from individual countries (to remove any discrepancies with the rounded total reported in raw Comext data).

<sup>(2)</sup> 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 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.

### 13.4.3 Article Production Stage

Regardless of the feedstock used (PP, LDPE, Bio-PP and Bio-LDPE), manufacturing of wipes has been assumed to take place via blown film extrusion of melted plastic granules. This process was chosen as an approximation of the melt-blown process described in the literature for the production of non-woven wipes.

The burdens of the overall conversion process were modelled through the aggregated, EF dataset "[EU-28+EFTA] Film Extrusion (blowing); plastic extrusion | production mix, at plant | for PP, PE, PVC, PET and PS", which accounts for a 99% conversion efficiency. The burdens of the respective disposal process were additionally included in the model, assuming that they are entirely sent to recycling, ultimately replacing virgin granulates of the same material. Hence, recycled PP granules were assumed to replace virgin, fossil-based PP granules, since no bio-based PP was reported to be available on the market at the time of developing this analysis (EUBP, 2019). Similarly, recycled LDPE granules (both fossil-based and bio-based) replaced virgin, fossil-based LDPE granules, being the estimated share of bio-based LDPE on the market very low (i.e. 0.2%). 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 PP and LDPE film, in the absence of more specific data for recycling of pre-consumer, industrial scraps. For further detail

on the modelling of recycling and avoided virgin material production (including the implementation of the Circular Footprint Formula), the reader is hence referred to Section 5.4.5.3 on food packaging film.

#### **13.4.4 Distribution Stage**

The transport of the wipes from the manufacturing site to the final user was modelled based on the default transport scenario specified in the PEF context (and in the Method) for the route *factory -> retail -> final client*. The following routes were thus considered:

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

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

#### **13.4.5 End of Life Stage**

This section describes the modelling of the End of Life stage of wipes. In particular, Section 13.4.5.1 focuses on the definition of the EU-average End of Life scenario, which is considered as a base case for the calculation of the potential impacts of the compared LCA scenarios. The remaining sections (13.4.5.2 – 13.4.5.4) address the modelling of waste collection and transport, and of the different End of Life options explored in the study. Finally, Section 13.4.5.5 provides case-specific details on the estimate of the potential contribution of wipes to macro-plastic formation at the End of Life (including product litter) and micro-plastic generation throughout the supply chain.

##### **13.4.5.1 End of Life scenario**

Concerning the End of Life of wipes, no case-specific data were found in the literature apart from the fact that wipes are not collected separately for recycling. Therefore, for the establishment of the average End of Life scenario, the EU average rates regarding the fate of municipal solid waste were used i.e. 53% incineration and 47% landfilling. In the absence of any specific data, it was deemed reasonable to assume the same landfill and incineration rate in all scenarios. Nevertheless, it is highly likely that the End of Life is defined by the functionality of the plastic article and not by the feedstock from which it is produced.

##### **13.4.5.2 Modelling of waste collection and transport**

Relevant data for the modelling of collection and transport of non-separately collected waste wipes to incineration and landfilling (as residual waste) were taken from Rigamonti et al. (2013), and implemented in the model as detailed in Section 4.4.5.2. When EF incineration and landfilling datasets were used, no transport of collected waste was modelled separately, since the respective burdens are already accounted for in such datasets. In this case, only collection was thus modelled, according to the approach described in Section 4.4.5.2.

##### **13.4.5.3 Modelling of incineration**

For conventional fossil-based polymers (i.e. PP and LDPE) aggregated, material-specific incineration datasets are available from the EF database, and were applied to model the treatment of pots made of these materials in a municipal waste incineration plant. Similarly, for bio-based PP and bio-based LDPE wipes, partially aggregated, material-

specific inventories from the GaBi database were applied (no EF datasets are available for these polymers).

All the selected datasets are developed based on a waste-specific incineration model, which has been described more in detail in Section 4.4.5.4. 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 general approach to handle energy recovery situations adopted in the EF context and in the Method (Report I), the product system generating the waste material sent to incineration (i.e. the wipes life cycle, in this case) takes the full burdens from the incineration process. However, it is credited for 100% of the benefits from avoided production of conventional energy (electricity and heat) replaced by energy recovered from waste. While in EF incineration datasets these credits are already accounted for in the aggregated inventory, for GaBi datasets they were added to the main process inventory. Therefore, the EU residual electricity grid mix (as modelled in the EF 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 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 includes 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. Also, thermal energy from waste (11%) was excluded, as according to the approach described above for the handling of energy recovery situations, 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).

#### **13.4.5.4 Modelling of landfilling**

Landfilling of wipes made of conventional, non-biodegradable polymers (i.e. fossil-based PP and LDPE) was modelled based on a common, aggregated EF dataset representing disposal of plastic waste in a managed, municipal waste landfill ([EU-28+EFTA] Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific sites)). The underlying inventory is material-specific, but refers to the average chemical composition and degradability values of generic plastic waste, rather than to those of the specific polymer. 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<sup>223</sup>. 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 emissions to the environment over a 100 year time horizon. Emissions occurring beyond 100 years are not accounted for in the model. Landfill gas generation is calculated based on the organic carbon content in the waste and the respective degradation rate assumed over 100 years (not reported), while, for simplification reasons, an average landfill gas composition for the stable methane phase

<sup>223</sup> While more polymer-specific landfilling datasets are available from other databases (i.e. ecoinvent), the mentioned EF dataset for landfilling of generic plastic was selected, as specifically referring to EU as the reference geography (in contrast to available polymer-specific datasets), and to keep consistency with the overall dataset selection "hierarchy" followed in this project (Section 3.5).

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 rate of landfill gas, energy efficiencies of engines, collection rate of leachate and respective treatment efficiencies). Further details on how these parameters were set to reflect the EU-average situation are provided in Section 4.4.5.5.

The same dataset described above for landfilling of generic plastic waste was also employed as a proxy for landfilling of pots made of “drop-in”, non-biodegradable, bio-based polymers, i.e. bio-based PP and bio-based LDPE. Compared to the original dataset, emissions of CO<sub>2</sub> and CH<sub>4</sub> 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 degradation and those coming from background activities, the entire amounts of fossil CO<sub>2</sub> and CH<sub>4</sub> reported as released to air were converted to biogenic emissions. This approximation is considered acceptable, as reported CO<sub>2</sub> and CH<sub>4</sub> emissions are modest, and overall amounting to only 1.4% of the carbon content in the landfilled plastic material, despite they also include the contribution of emissions from background processes. This can be partly explained by the generally low degradation (and mineralisation) of conventional, non-biodegradable polymers in landfill, which is typically assumed to be in the range of 1% over 100 years (Doka, 2009b). On the other hand, it is acknowledged that this approach is in favour of the two bio-based polymers (i.e. Bio-PP and Bio-LDPE).

In line with the time horizon applied for landfill emission modelling in the selected dataset, (biogenic) carbon in polymers that is not degraded (mineralised) after 100 years from deposition was considered to be never released from the landfill body. As a base case, this missed release of atmospheric carbon taken up during biomass growth is not reflected in Climate Change LCIA results for bio-based PP and LDPE wipes, since characterisation factors for biogenic CO<sub>2</sub> flows have been set to zero (fully conforming to prescriptions in the PEF method). However, to show the relevance of this choice on the overall results, the Climate Change impact of the two bio-based alternatives was also calculated by acknowledging the effects of non-released biogenic carbon. For this purpose, a specific CO<sub>2</sub> uptake was modelled in the inventory, expressing the net amount of CO<sub>2</sub> taken up during biomass growth and not released (as CO<sub>2</sub> or CH<sub>4</sub>) during polymer degradation in landfill. The uptake was calculated based on the biogenic carbon content in the polymer, and assuming a mineralisation rate over 100 years equal to 1% for both polymers (Doka, 2009b). While the mineralisation rate assumed in the dataset is not reported explicitly, it seems to be in line with this assumption (according to the figures reported above regarding the inventoried carbon emissions). Considering a biogenic carbon content equal to 81.9% for both Bio-PP and Bio-LDPE, a carbon uptake equal to 2.97 kg CO<sub>2</sub>/kg bio-based polymer was calculated. This uptake was then characterised with a factor of -1 kg CO<sub>2</sub> eq. per kg CO<sub>2</sub>-C not degraded.

#### **13.4.5.5 Contribution to macro- and micro-plastics generation (including product litter)**

The contribution to macro- and micro-plastics loss and release of the analysed wipes scenarios was estimated according to the framework outlined in Annex B of the Method (Report I), and summarised in Section 4.4.5.6 of this report in terms of underlying quantification methods and approaches. In this section, the case-specific parameters considered to apply the latter to the assessed wipes LCA scenarios are reported.

##### **13.4.5.5.1 PLP method**

Regarding the estimate of the loss and release of macro-plastics at the End of Life stage (from product littering and waste mismanagement), the product-specific parameters reported in Table 13.6 were considered to apply the first level of the framework (“PLP method”) to wipes (derived from Peano et al., 2020). Note that such parameters are

defined regardless of the type of feedstock or material used for manufacturing. Similarly, all wipes scenarios contribute to macro-plastics loss and release to the same extent, as the chosen feedstock or material do not affect the probability of the article to be littered.

**Table 13.6.** Case-specific parameters of the *PLP method* applied to the wipes LCA scenarios.

Parameter ( <sup>1</sup> )	Value
Littering rate (LR <sub>lit</sub> ) (%)	2
Release rate to ocean (Rel <sub>ocean</sub> ) (%)	25
Release rate to the terrestrial environment (Rel <sub>terenv</sub> ) (%)	75

(<sup>1</sup>) For details on the meaning of each parameter, refer to Annex B of the Method (Report I).

No case-specific parameters are required to calculate the supply-chain loss and release of micro-plastics to ocean and to the terrestrial environment by applying the PLP method. Indeed, the contribution of the considered micro-plastics sources (pellet loss and tire abrasion) depends on the magnitude of the relevant life cycle processes (i.e. polymer production and road transport) in the specific wipes scenario (which in turns depend on the reference flow<sup>224</sup> of each scenario). However, the quantification is made by means of default parameters that are not affected by the type of product, polymer or feedstock source (as reported in Annex B of the Method (Report I)).

#### 13.4.5.5.2 Expanded PLP method

The parameters required to calculate the micro-plastics loss and release from marine coatings and road markings are not case-specific, and are reported in Annex B of the Method (Report I). Similar to the other considered micro-plastics sources, the contribution of these additional sources depends on the intensity of relevant (transport) processes in the specific product inventory, by means of the above-mentioned case-unspecific parameters.

As for the calculation of the additional contribution of plastic waste exported for recycling to the loss and release of macro-plastics from waste mismanagement, this case study considers that there is no recycling stream at the End of Life of the product. Thereby, no additional calculation due to exported plastic waste for recycling has been performed.

### 13.4.6 Calculation of iLUC impacts

As a base case, the contribution of Indirect Land Use Change (iLUC) to the potential Climate Change impact of bio-based wipes alternatives was calculated according to the approach outlined in Section 4.4.16.4 of the Method (Report I).

In order to apply (recalculated) iLUC GHG emission factors from the EU 2015/1513 Directive (EC, 2015), the specific land demand for the production of relevant crops (m<sup>2</sup>\*year / kg crop) was calculated first. The calculation was based on the aggregated amount of arable and agriculture land occupation flows reported, for the relevant country (e.g. Brazil for sugarcane), in the dataset used to model the production of the specific crop. If the geography of such flows was not specified, all the arable and agricultural land occupation flows reported in the dataset were aggregated. These estimates were checked against values of land demand calculated based on 5-years average crop yields from FAOSTAT (FAO, 2019), which were found to be generally aligned with the former (absolute variation between 3% and 20%). Hence, the values estimated based on land occupation flows were applied, for consistency with the overall LCI modelling of scenarios.

Specific land demand for crop production was then converted into a demand per functional unit, based on the specific consumption of crop for polymer production (kg crop/kg polymer, consistently with the values applied in the LCI modelling) and the

<sup>224</sup> The reference flow is the mass of wipe material required to fulfil the functional unit.

- 1 amount of polymer needed to fulfil the functional unit (reference flow) in the relevant
- 2 scenario. The iLUC contribution to the Climate Change impact was finally calculated by
- 3 applying to the latter the recalculated GHG emission factors from the Directive. The
- 4 described calculation steps to estimate the iLUC contribution to the potential Climate
- 5 Change impact are summarised in Table 13.7.

1 **Table 13.7** Calculation of the iLUC contribution to the potential Climate Change impact of wipes LCA scenarios relying on bio-based polymers.

Scenario / Polymer	Feedstock	Land demand for crop production [m <sup>2</sup> *y/kg <sub>crop</sub> ]	Crop demand for polymer production [kg <sub>crop</sub> /kg <sub>polymer</sub> ]	Polymer demand per F.U. [kg <sub>polymer</sub> /FU]	iLUC GHG emission factor [kg CO <sub>2</sub> eq./ (m <sup>2</sup> *y)]	iLUC Climate Change impact [kg CO <sub>2</sub> eq./FU]
Bio-PP	Sugarcane (BR)	0.117	25.7	2.47	0.176	1.31
Bio-LDPE	Sugarcane (BR)	0.117	23.8	2.47	0.176	1.21

## 13.5 Life Cycle Impact Assessment results

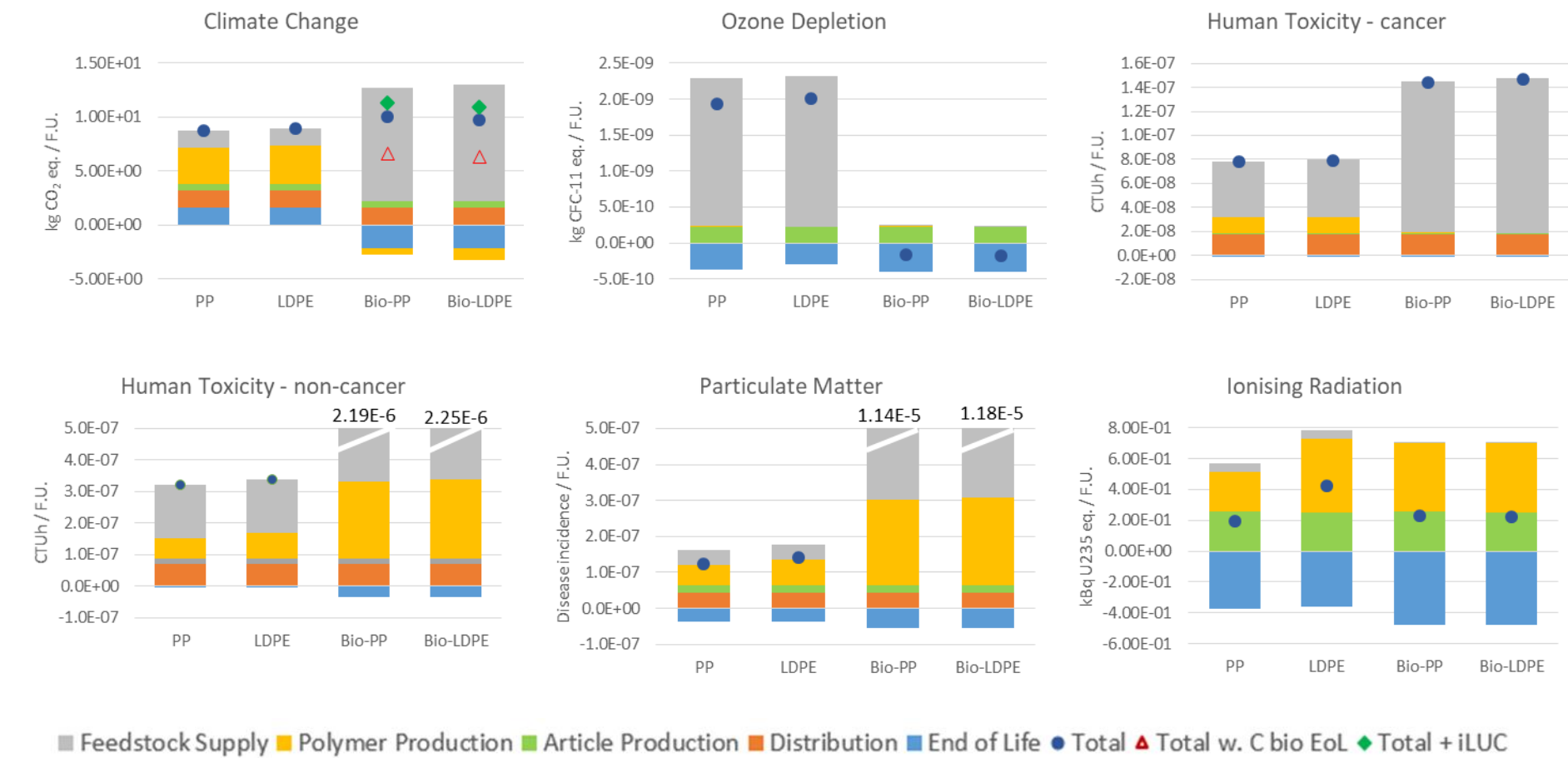
The characterised potential impacts of the examined scenarios (with and without the iLUC contribution to Climate Change) are reported in Figures 13.5 to 13.7. These also show the breakdown of contributions from the main lifecycle stages, which include:

- i) *Feedstock Supply*, i.e. oil/natural gas extraction, transport and possible refining, transport of naphtha from refinery to downstream user; or crop cultivation and transport to further processing (bio-based polymers);
- ii) *Polymer Production*, i.e. all gate-to-gate activities from feedstock processing to monomer/intermediate production and polymerisation, including any transport among these as well as transport of polymer granulate to wipes manufacturing site;
- iii) *Article production*, i.e. conversion of the polymer into wipes by stretch-blow moulding;
- iv) *Distribution*, i.e. transport of wipes from the manufacturing site to the final user; and
- v) *End of Life*, i.e. waste wipes collection, transport and treatment or disposal, as well as any avoided processes from downstream displacement of energy.

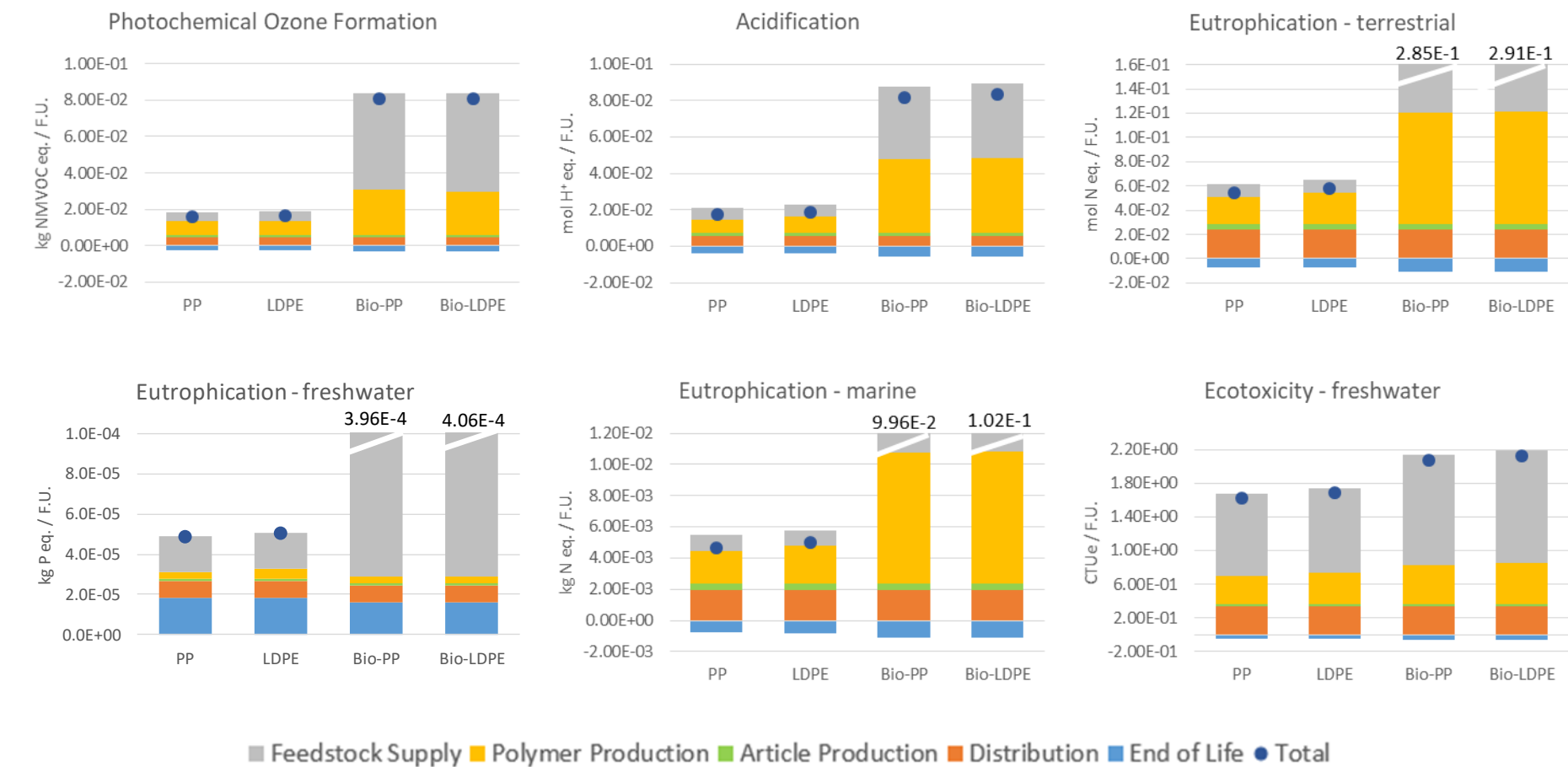
Figure 13.8 shows the contribution of each sub-category to the Climate Change impact (i.e. from fossil, biogenic and land use/land transformation GHG emissions), while the overall characterised, normalised and weighted impact assessment results are reported in Tables B.10.5 to B.10.7 in Annex B.10. The latter are not reported in this section to keep readability of the document, despite this is not in alignment with the position suggested in the LCA report template provided in the Method (Report I).

Note that scenario impacts presented in Figures 13.5 to 13.7 refer to the EU-average End of Life scenario (as described in Section 13.4.5.1), and represent the net impacts from waste management activities, resulting from the balance between real burdens and benefits (if any). Potential impacts calculated assuming 100% of post-consumer wipes being routed to each viable End of Life option are presented in Figures 13.9 to 13.11. We warn that this analysis should not be interpreted as a direct LCA comparison of alternative End of Life management options for wipes, because a product-based perspective is considered, and allocations among consecutive product life cycles are applied (e.g. in case of recycling).

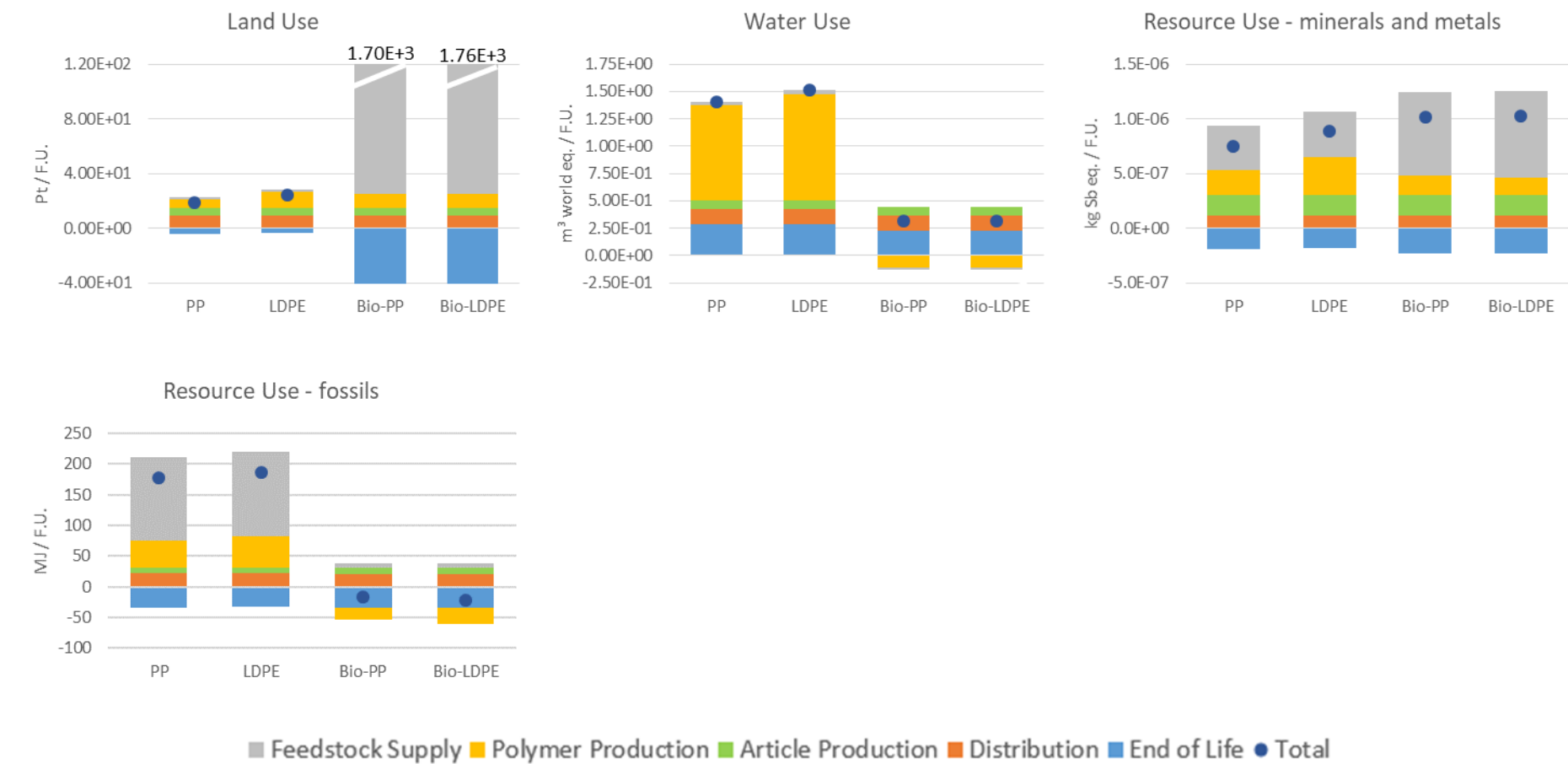




**Figure 13.5.** Potential impact of wipes LCA scenarios for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity - non-cancer, Particulate Matter and Ionising Radiation. Note that in some impact categories a part of the results is out of scale and is curtailed. Climate Change impacts denoted with "C bio EoL" accounts for the contribution of biogenic carbon not released after 100 years from landfilling of bio-based wipes.

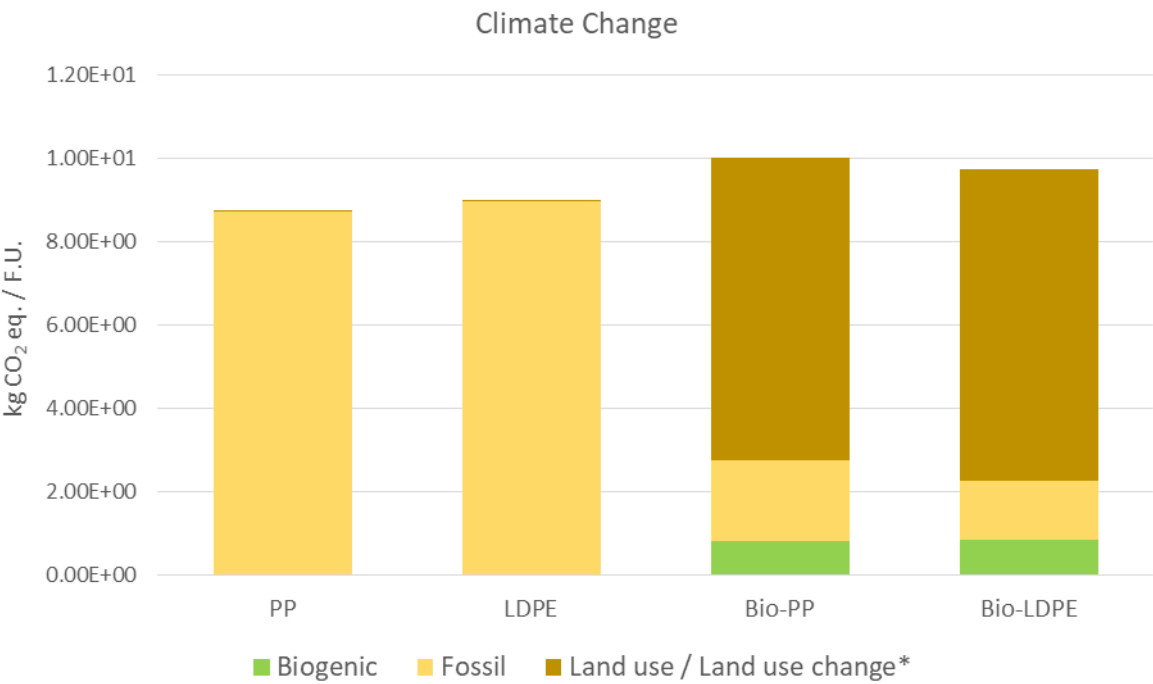


**Figure 13.6.** Potential impact of wipes LCA scenarios for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity – freshwater. Note that in some impact categories a part of the results is out of scale and is curtailed.



**Figure 13.7.** Potential impact of wipes LCA scenarios for different End of Life options, for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils. Note that in some impact categories a part of the results is out of scale and is curtailed.

1

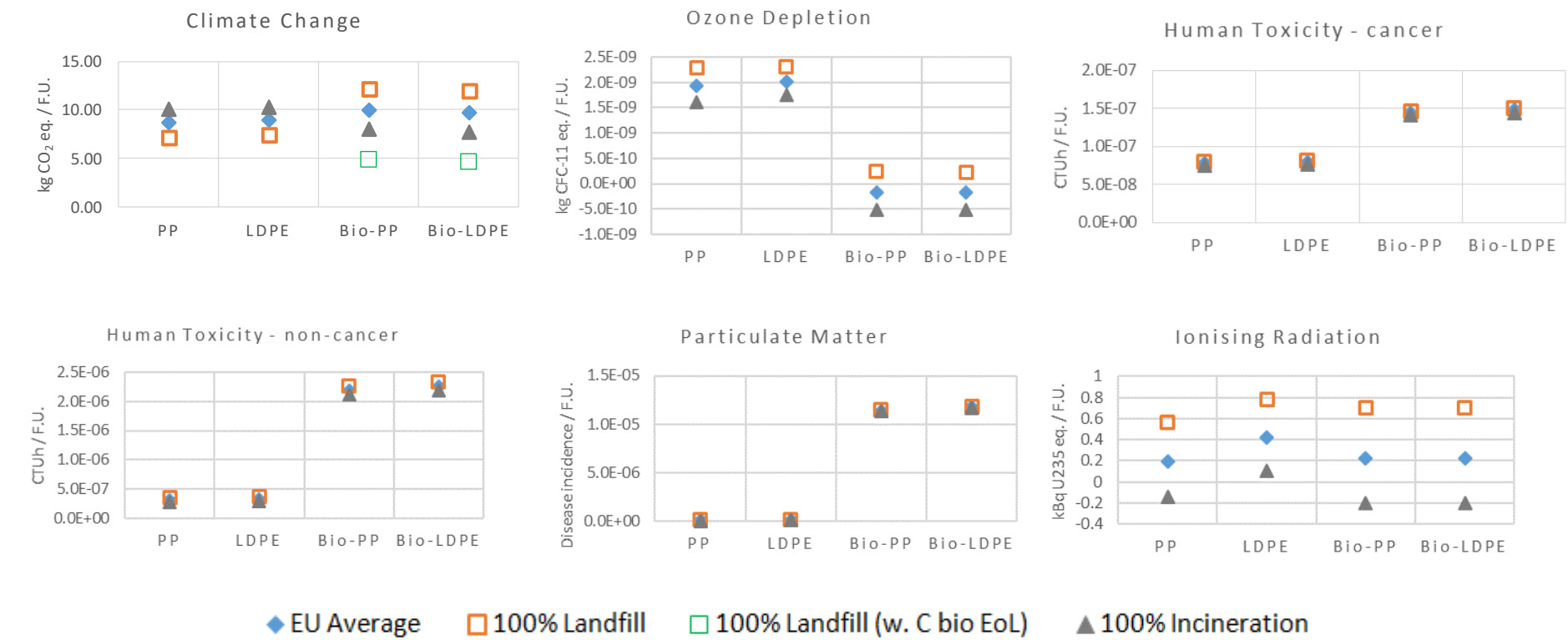


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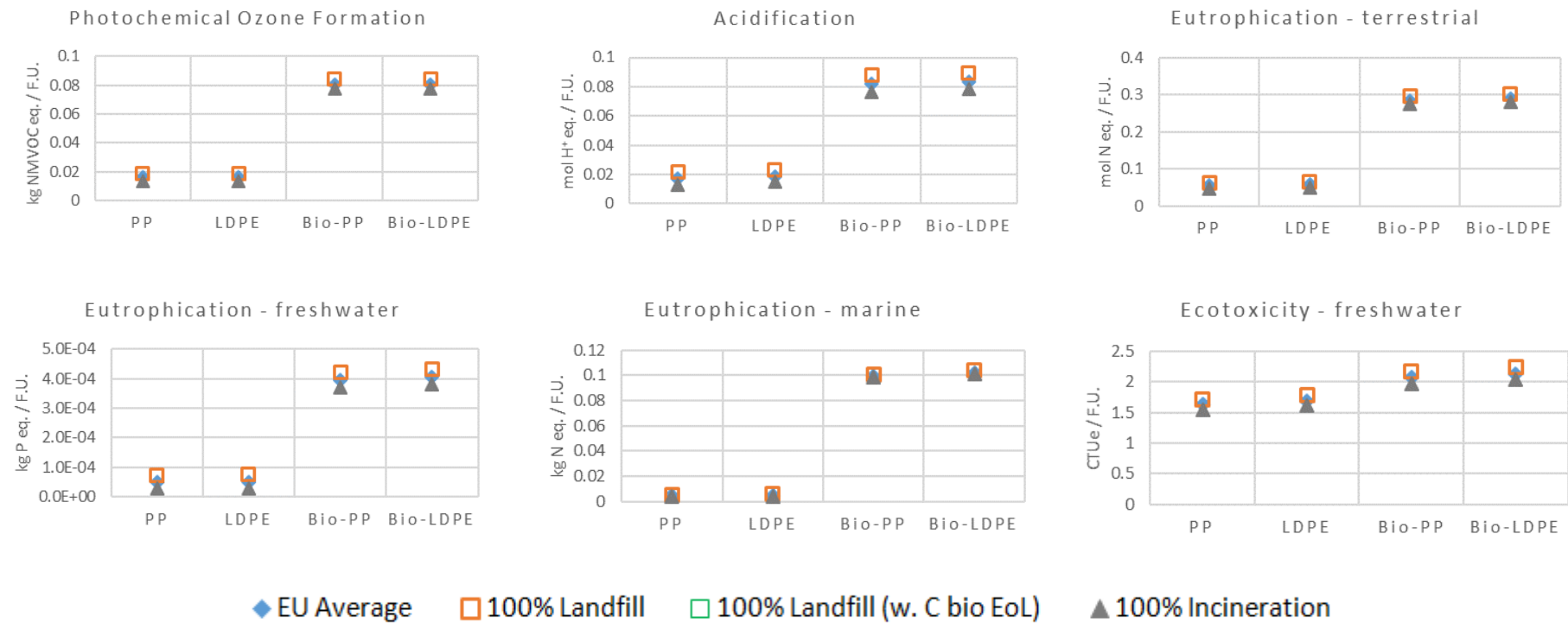
3 **Figure 13.8.** Contribution of fossil, biogenic, and land use/land transformation GHG emissions to  
4 the total Climate Change impact of wipes LCA scenarios (note that the contribution of iLUC to land  
5 transformation emissions is excluded).

6

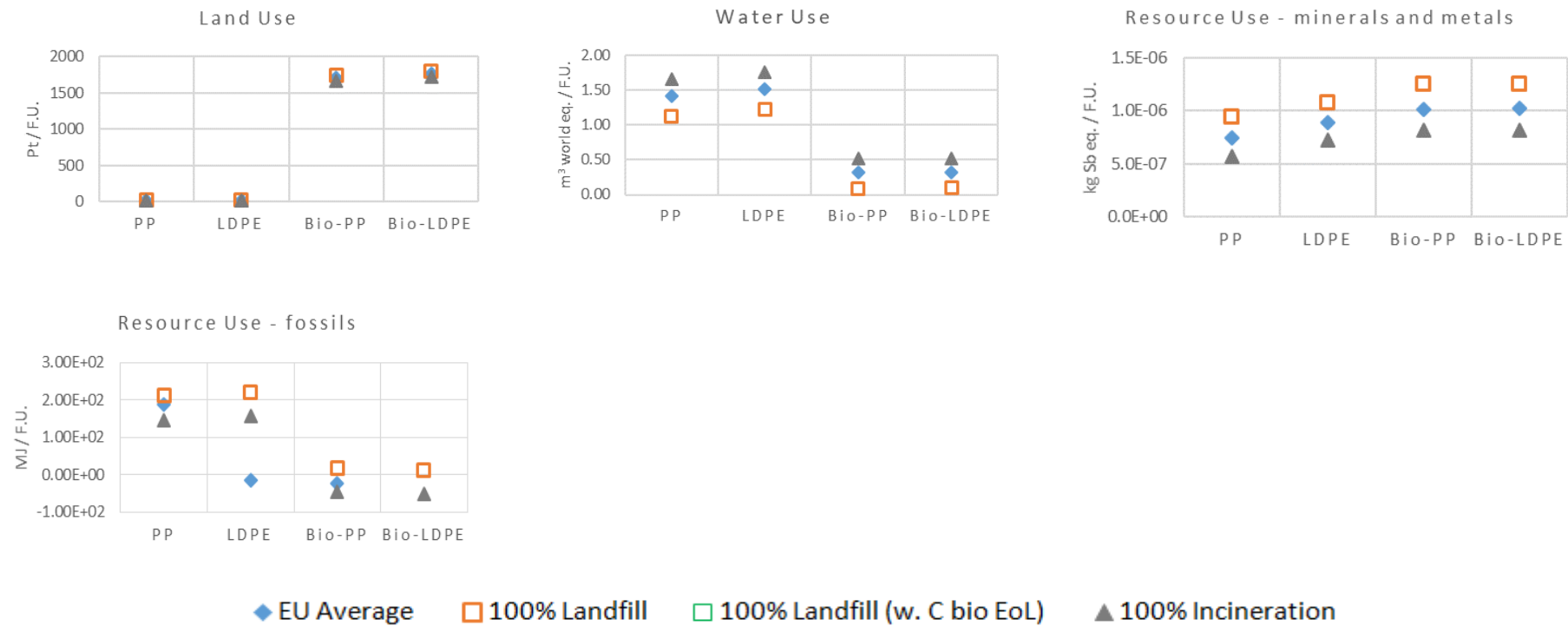
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**Figure 13.9.** Potential impact of wipes LCA scenarios for different End of Life options, for the categories of Climate Change, Ozone Depletion, Human Toxicity - cancer, Human Toxicity – non-cancer, Particulate Matter and Ionising Radiation. The Climate Change impacts of 100% landfilling scenario denoted with “C bio EoL” accounts for the contribution of biogenic carbon not released after 100 years from landfill deposition of bio-based wipes.



**Figure 13.10.** Potential impact of wiper LCA scenarios for different End of Life options, for the categories of Photochemical Ozone Formation, Acidification, Eutrophication - terrestrial, Eutrophication - freshwater, Eutrophication - marine and Ecotoxicity - freshwater



**Figure 13.11.** Potential impact of wipes LCA scenarios for different End of Life options, for the categories of Land Use, Water Use, Resource Use - minerals and metals, Resource Use – fossils.

## 13.6 Additional Environmental Information

This section presents additional environmental impacts or aspects going beyond the default impact categories considered in the present method, which reflect official categories adopted in the PEF context. Additional environmental issues addressed include indirect Land Use Change (iLUC) impact on Climate Change, potential Biodiversity impacts, as well as the contribution of the investigated wipes scenarios to macro-plastics formation at End of Life (including product littering) and to micro-plastics generation throughout the supply chain.

### 13.6.1 iLUC impact

Table 13.8 presents the estimated contribution of GHG emissions from iLUC induced by bio-based feedstock supply to the potential Climate Change impact of bio-based wipes.

**Table 13.8.** iLUC contribution to the potential Climate Change impact of wipes LCA scenarios.

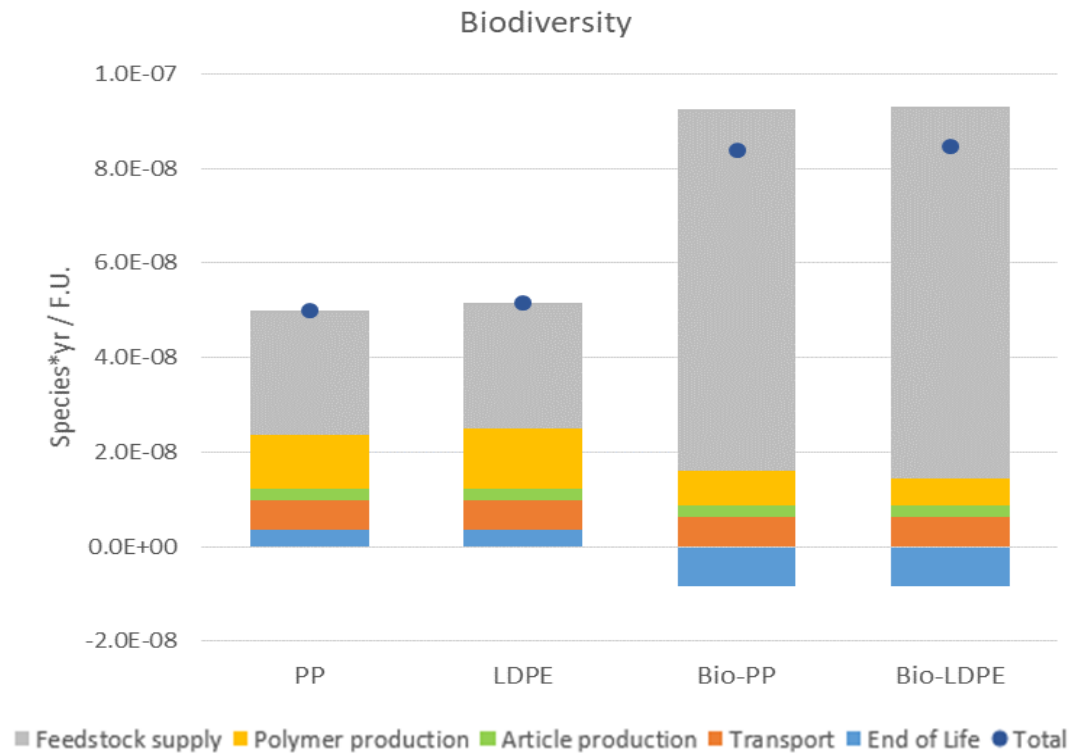
Scenario / Polymer	iLUC Climate Change impact [kg CO <sub>2</sub> eq/FU]	Total Climate Change impact (incl. iLUC) <sup>(1)</sup> [kg CO <sub>2</sub> eq/FU]
S1 - PP	-	(8.7)
S2 – LDPE	-	(9.0)
S3 – Bio-PP	1.31	11.3 (10.0)
S4 – Bio-LDPE	1.21	10.9 (9.7)

<sup>(1)</sup> Values in parenthesis refer to the Climate Change impact without the iLUC contribution.

### 13.6.2 Biodiversity impacts

Estimated potential Biodiversity impacts of the investigated scenarios are presented in Figure 13.12. It is reminded that the assessment is performed through an endpoint-level indicator relying on impact assessment methods that for some of the underlying midpoint impact drivers differ from those recommended in the present method.

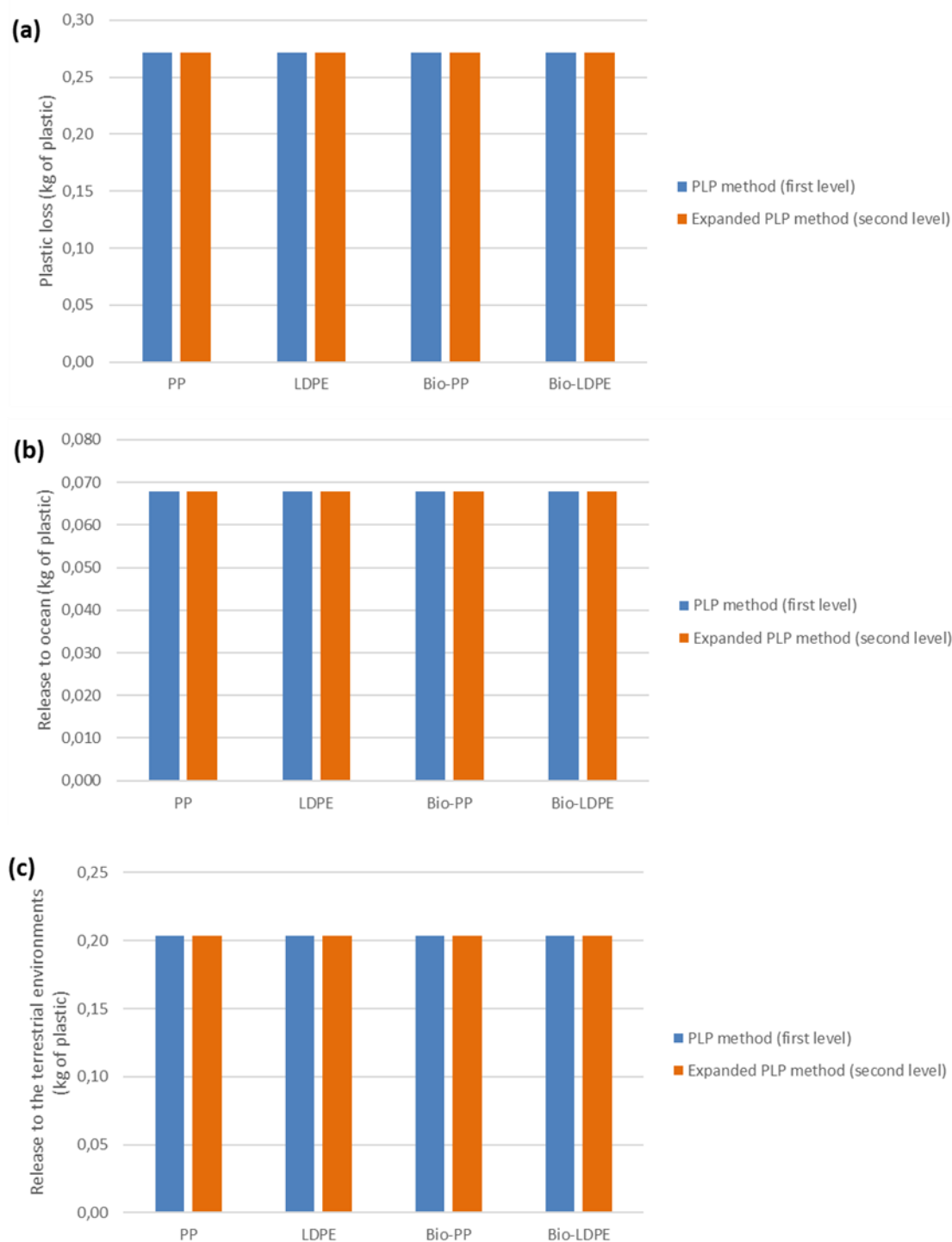




**Figure 13.12.** Potential biodiversity impact of wipes LCA scenarios, expressed as potential loss of species per year.

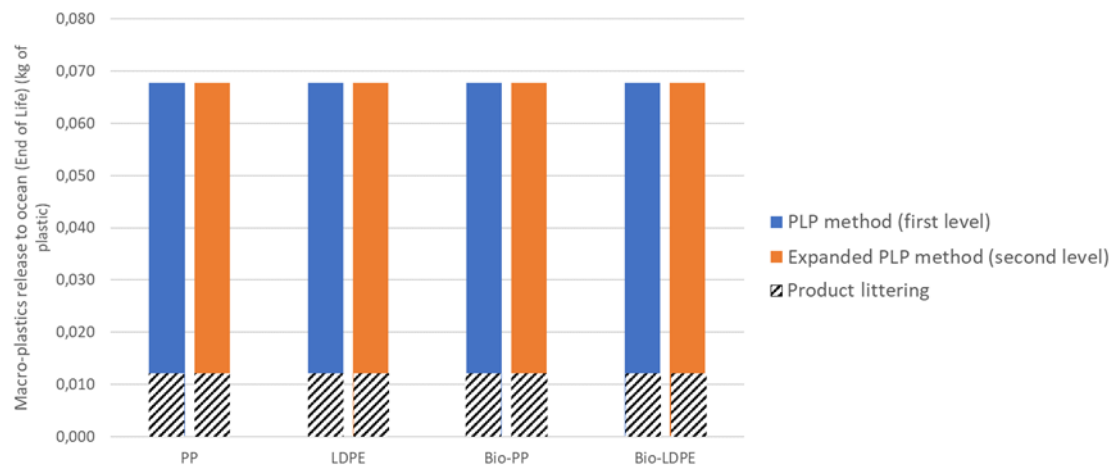
### 13.6.3 Macro- and micro-plastics generation

Figure 13.13 shows the total plastic loss, release to ocean and release to the terrestrial environment estimated for the assessed wipes scenarios, considering the whole product life cycle. Estimates obtained by applying all the two different explored approaches are presented, including the total loss and release to ocean/terrestrial environment calculated through both the PLP method (first level) and the Expanded PLP method (second level).

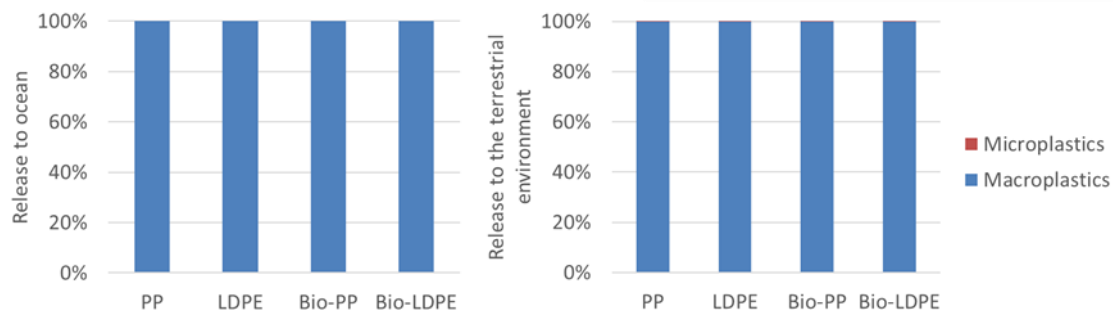


**Figure 13.13.** Total plastic loss (a), release to ocean (b) and release to the terrestrial environment (c) estimated for wipes LCA scenarios with different approaches. The contribution of both macro- and micro-plastics is included, considering the whole product life cycle.

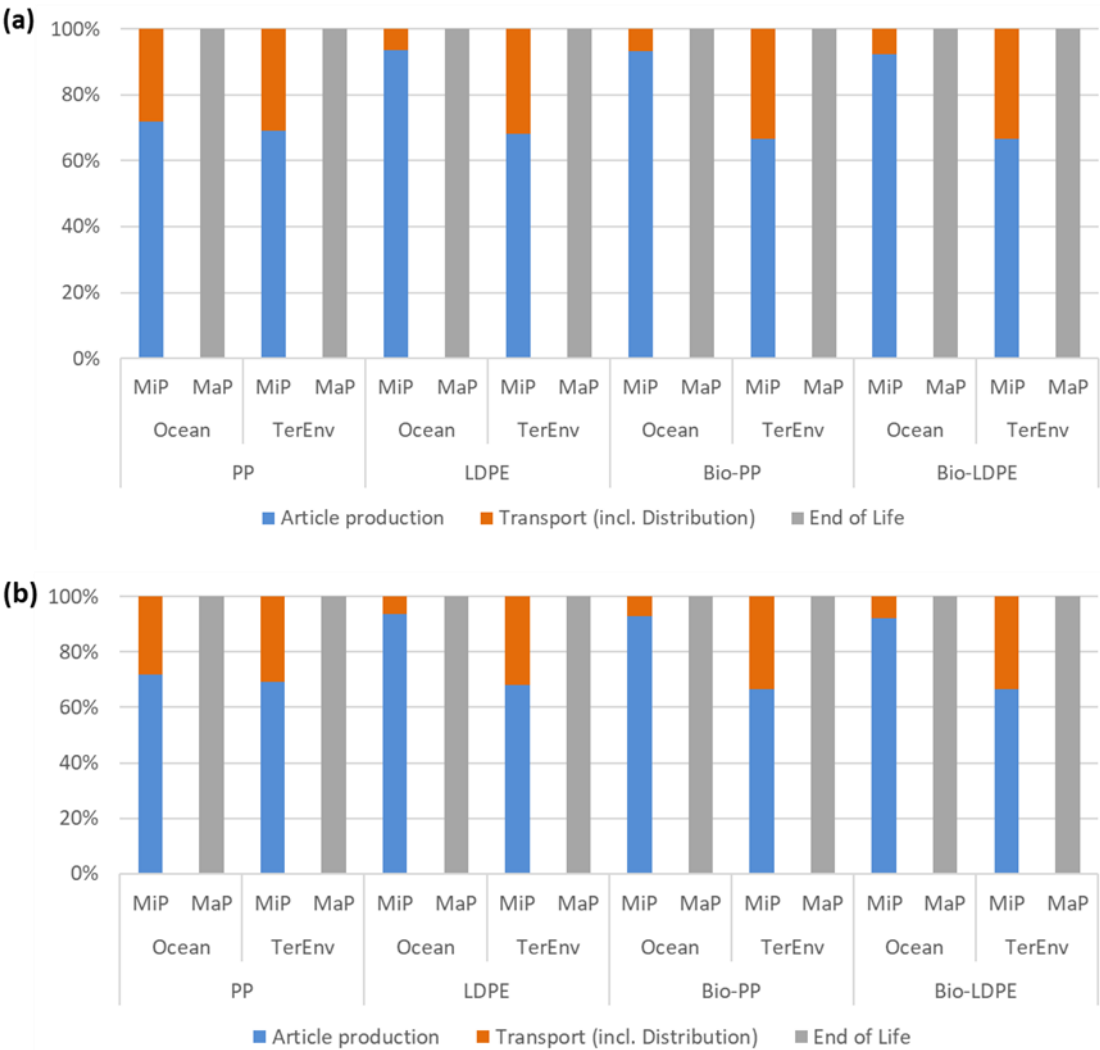
Regarding the sole release of macro-plastics to ocean at End of Life (from product littering and waste mismanagement), Figure 13.14 presents the contribution of each wipes scenario, estimated with the two different explored approaches: the PLP method (first level), the Expanded PLP method (second level).



**Figure 13.14.** Macro-plastics release to ocean at End of Life, estimated with different approaches. As for the share between the release of macro- and micro-plastics, Figure 13.15 shows as an example the relative contribution of the two categories to the total release to ocean and to the terrestrial environment estimated through the PLP method (first level). Note that the results are similar to those obtained when applying the Expanded PLP method (second level), as further discussed in Section 13.7.7.



**Figure 13.15.** Contribution of macro- and micro-plastics to the total plastic release to ocean and to the terrestrial environment estimated with the PLP method for each wipes LCA scenario. The contribution of the different lifecycle stages to the total macro- and micro-plastics release to ocean and to the terrestrial environment of the assessed wipes scenarios has been finally evaluated. The results of this “hotspot analysis” are reported in Figure 13.16 for both the PLP method and the Expanded PLP method.



**Figure 13.16.** Contribution of life cycle stages to the total macro-plastics (MaP) and micro-plastics (MiP) release to ocean (Ocean) and to the terrestrial environment (TerEnv) estimated for wipes LCA scenarios with (a) the PLP method (first level) and (b) the Expanded PLP method (second level).

### 13.7 Interpretation

In the interpretation of case study results, most relevant impact categories and lifecycle stages of the studied systems are firstly identified (13.7.1 and 13.7.2, respectively). Scenario impacts are then compared, considering both characterised LCIA results (13.7.3) and total normalised and weighted impacts (13.7.4). Results obtained by applying individually each viable End of Life option are separately discussed in Section 13.7.6, while the effects of iLUC and the contribution to macro- and micro-plastics generation are addressed in Sections 13.7.5 and 13.7.7, respectively.

Most relevant processes were not identified, since it was not possible to achieve a similar level of vertical disaggregation for foreground processes included in the life cycle models of the analysed scenarios. Therefore, the identification of most relevant processes cannot be carried out consistently across all scenarios, and a more detailed investigation for specific scenarios would not be meaningful. The identification of most relevant elementary flows was also not undertaken, as this would require prior identification of most relevant processes. Note, however, that any company, organisation or any other supply chain actor applying the present method shall proceed with the identification of both most relevant processes and elementary flows.

### 13.7.1 Identification of most relevant impact categories

Table 13.9 shows the most relevant impact categories identified for each wipes scenario, based on normalised and weighted impacts, according to the approach outlined in the Method (Report I, Section 6.2.1). Relevant categories were identified as those that cumulatively contribute to at least 80% of the total normalised and weighted impact of the considered impact categories, excluding toxicity-related ones (these being still based on characterisation factors implemented in the EF 2.0 impact assessment methods, and hence not yet updated based on REACH data<sup>225</sup>). 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 were identified as the two most relevant categories in the fossil-based wipes (PP and LDPE), followed by Acidification in the lower share. On the other hand, Particulate Matter was identified as the most relevant category for the bio-based wipes (Bio-PP and Bio-LDPE), followed by Climate Change and Eutrophication – marine.

**Table 13.9.** Most relevant impact categories identified for wipes LCA scenarios and related contribution to the total normalised and weighted scenario impact.

S1 – Fossil-based PP		S2 – Fossil-based LDPE	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Climate Change	42.7%	Climate Change	41.7%
Resource Use - fossils	41.4%	Resource Use - fossils	41.5%
Acidification	3.5%	Acidification	3.7%
<b>Total</b>	<b>87.6%</b>	<b>Total</b>	<b>86.9%</b>
S3 – Bio-based PP		S4 – Bio-based LDPE	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Particulate Matter	69.4%	Particulate Matter	70.2%
Climate Change	11.6%	Climate Change	11.1%
Eutrophication - marine	4.5%	Eutrophication - marine	4.5%
<b>Total</b>	<b>85.4%</b>	<b>Total</b>	<b>85.8%</b>

### 13.7.2 Identification of most relevant life-cycle stages

Table 13.10 shows the most relevant lifecycle stages in the relevant impact categories identified in Section 13.7.1, and the associated contribution, quantified according to the rules detailed in the Method (Report I, Section 6.2.2). Most relevant stages include those that together contribute to at least 80% of the total lifecycle impact in the specific category, and are highlighted in yellow in Table 13.10. For a more exhaustive picture, the contribution of all lifecycle stages is reported.

For the fossil-based wipes (PP and LDPE), Polymer Production is identified as the most relevant stage for the Climate Change (with a contribution of 38% for PP and 40% for

<sup>225</sup> According to the latest version of the Product Environmental Footprint Category Rules Guidance (EC, 2018), toxicity-related impact indicators based on these characterisation factors shall be excluded from the procedure to identify most relevant impact categories.

LDPE wipes) whereas End of Life, Feedstock Supply and Distribution are following with roughly 18% each for both polymers. Concerning the Resource Use - fossils, Feedstock Supply is contributing 76% and 73% of the total burden in the case of PP and LDPE wipes, respectively.

The picture is different in the case of the bio-based wipes. In both cases assessed (Bio-PP and Bio-LDPE), almost the total burden comes from the Feedstock Supply. More precisely this life cycle stage contributes 98% of the burden in the Particulate Matter. Regarding the Climate Change, the contribution of the Feedstock Supply is 105% for the Bio-PP wipes and 111% for the Bio-LDPE wipes.

**Table 13.10.** Contribution of life cycle stages to the total characterised impacts of the most relevant categories identified for wipes LCA scenarios. Most relevant stages (i.e. those contributing to more than 80% of the total impact) are highlighted in yellow.

<b>S1 – Fossil-based PP</b>					
<b>Climate Change</b>		<b>Resource Use - fossils</b>		<b>Acidification</b>	
<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%	Feedstock Supply	76%	Polymer Production	41.4%
End of Life	19%	Polymer Production	26%	Feedstock Supply	37.4%
Feedstock Supply	18%	Distribution	12%	Distribution	31.7%
Distribution	18%	Article Production	6%	Article Production	11.8%
Article Production	7%	End of Life	-19%	End of Life	-22.3%
<b>S2 – Fossil-based LDPE</b>					
<b>Climate Change</b>		<b>Resource use - fossils</b>		<b>Acidification</b>	
<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	40%	Feedstock Supply	73%	Polymer Production	46.0%
Feedstock Supply	18%	Polymer Production	28%	Feedstock Supply	34.8%
End of Life	18%	Distribution	11%	Distribution	29.0%
Distribution	18%	Article Production	6%	Article Production	10.8%
Article Production	7%	End of Life	-18%	End of Life	-20.6%
<b>S3 – Bio-based PP</b>					
<b>Particulate Matter</b>		<b>Climate Change</b>		<b>Eutrophication - marine</b>	
<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	98%	Feedstock Supply	105%	Feedstock Supply	90.3%
Polymer Production	2%	Distribution	16%	Polymer Production	8.4%
Distribution	0%	Article Production	6%	Distribution	1.9%
Article Production	0%	Polymer Production	-5%	Article Production	0.5%
End of Life	-1%	End of Life	-22%	End of Life	-1.1%

<b>S4 – Bio-based LDPE</b>					
<b>Particulate Matter</b>		<b>Climate Change</b>		<b>Eutrophication - marine</b>	
<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	98%	Feedstock Supply	111%	Feedstock Supply	90.5%
Polymer Production	2%	Distribution	16%	Polymer Production	8.3%
Distribution	0%	Article Production	6%	Distribution	1.9%
Article Production	0%	Polymer Production	-12%	Article Production	0.4%
End of Life	-1%	End of Life	-22%	End of Life	-1.1%

### 13.7.3 Interpretation of characterised results

In this section, the characterised potential impacts of the assessed scenarios (Figures 13.5-13.7 and Table B.10.5 in Annex B.10) are compared to evaluate the effects of using an alternative feedstock sources or material for a specific fossil-based polymer of reference used in wipes manufacturing. Therefore, the impacts of bio-based PP wipes are compared with those of fossil-based PP wipes, and bio-based LDPE wipes are compared to fossil-based LDPE wipes. In addition, the impacts of reference scenarios (fossil-based PP and LDPE) are initially compared, to provide useful elements to potentially extend the comparative considerations above provided below to any of such reference scenarios.

The comparison focuses on the sixteen, default impact categories considered in the Product Environmental Footprint (PEF) context and in this assessment. Potential impacts on Biodiversity are discussed separately, as being estimated via an endpoint indicator that is not recommended in the PEF framework, and partially relying on different impact assessment methods for the underlying midpoint-level impact categories. In the following comparison, differences between scenario impacts lower than 10% were not considered significant, in light of the uncertainty associated with the developed life cycle models and the applied impact assessment models. Moreover, any comparative considerations related to the Human Toxicity and Ecotoxicity impact categories need to be interpreted in light of the higher uncertainty of the underlying impact assessment models and results. The same applies, to a lower extent, also to Water Use and Land Use.

Focusing on the two fossil based-reference scenarios, no significant differences are observed between the PP and the LDPE wipes. The results in most of the impact categories assessed are comparable. However, in the Ionising Radiation, Land Use, Resource Use - mineral and metals and Particulate Matter, PP wipes show a better performance. More precisely, in the Ionising radiation, where the highest difference is observed, PP wipes perform 55% better than LDPE wipes.

Replacing fossil-based polymers (PP and LDPE) with their bio-based alternatives, is not proven to be beneficial for the assessed impact categories. On the contrary, the environmental impacts of Bio-PP and Bio-LDPE wipes are higher in most of the categories. Both bio-based alternatives perform, however, better in the categories: Ozone Depletion, Resource Use - fossils, Resource Use – mineral and metals, and Water Use. Bio-based LDPE is also better than the fossil-based LDPE in the categories Climate Change and Ionising Radiation. The reason of the generally worsened performance is to be searched in the high demand for inputs for the production of the feedstock required. A deeper investigation could not be performed, however, due to the aggregated nature of the LCI dataset used for the modelling of polymer production.

#### 13.7.4 Interpretation of normalised and weighted results

This section briefly compares the investigated scenarios based on the respective total impact scores, calculated by aggregating normalised and weighted impact assessment results across all impact categories (Table B.10.7 in Appendix B.10). The comparison is carried out according to the same criteria outlined in Section 4.7.3 in terms of contrasted wipes scenarios, and considering significant only impact differences larger than 10%. While affected by greater uncertainty (from the calculation and application of normalisation and weighting factors), total impact scores provides a more immediate and synthetic indication of the overall environmental performance of the analysed product scenarios. This facilitates the interpretation compared to looking at characterised results from several impact categories. However, such estimates are inevitably affected by value choices underlying the definition of weighting factors aimed at establishing an order of relevance for the different impact categories in a European decision context.

The comparison among total normalised and weighted scenario impacts reflects the overall picture of results emerged when focusing on characterised impact assessment results in single impact categories, and can be summarised in the following points:

- The two fossil-based scenarios (LDPE and PP) present a similar environmental impact.
- The bio-based scenarios (Bio-PP and Bio-LDPE) show a substantially worse performance than their fossil counterparts most probably because of the high land demand for the production of the relevant materials. More specifically, the impact of Bio-PP and Bio-LDPE is four times higher than fossil PP and LDPE, respectively

#### 13.7.5 Effects of indirect Land Use Change (iLUC)

When the contribution of GHG emissions from iLUC is taken into account, only a small changes are identified in the Climate Change impact of the bio-based alternatives (i.e. Bio-PP and Bio-LDPE wipes) (Figure 13.5 and Table 13.8). A maximum increase by 13% is observed for bio-based PP wipes, while for bio-based LDPE wipes the increase is equal to 12%.

Overall, accounting for iLUC effects does not affect the relative performance of the assessed scenarios (i.e. their comparison is not affected). However, this outcome needs to be read also in light of the GHG emission factors applied for iLUC in this assessment, which appear to fall in the lower end of the range of values available in the recent literature (see Section 4.4.16.3 in the Method (Report I)).

#### 13.7.6 Alternative End of Life Options

This section discusses characterised scenario results calculated by individually applying each viable End of Life option as an exclusive (100%) End of Life scenario (as presented in Figures 13.9 to 13.11). The main purpose is to evaluate how the performances of the single wipes scenarios are affected by the change in the applied End of Life option, rather than to mutually compare such scenarios for different End of Life options. However, the results should not be interpreted as a strict comparison among different End of Life alternatives for the studied article, since the evaluation applies a product perspective, and burdens/benefits of selected End of Life options (e.g. recycling) are partitioned between different product life cycles (which prevents from capturing the full implications of having a given waste stream routed to a specific End of Life option).

Because of the generally moderate contribution of the End of Life stage to the overall scenario impacts, the picture of results does not substantially change by considering 100% of a specific End of Life option rather than the EU-average End of Life scenario. No single End of Life option can be identified as preferable for all the impact categories assessed across the wipes scenarios. In general, according to the priority order outlined in the waste hierarchy, disposal is considered to be the least favourable option, and this is actually observed at least in some of the assessed scenarios. In the case of fossil-



based wipes, the impact categories that are negatively influenced when landfilling is chosen as the only End of Life option are Ozone Depletion, Human Toxicity – non-cancer, Particulate Matter, Ionising Radiation, Photochemical Ozone Formation, Acidification, the three categories related to Eutrophication, Land Use, Resource Use - mineral and metals and Resource Use - fossils. On the other hand, again considering the fossil-based wipes, incineration shows the worst performance in terms of Climate Change (and also Water Use) for obvious reasons. In the case of the bio-based alternatives, significant deviations from the EU average scenario are observed in few categories when the 100% End of Life options are assessed. In the frame of the present study, a deviation of more than 10% is characterised as significant. When 100% landfilling is selected, bio-based alternatives perform better in terms of Water Use. Moreover, landfilling seems the preferable End of Life option for the bio-based alternatives when the contribution of non-released biogenic carbon after 100 years of landfilling is taken into account. On the other hand, when 100% incineration is considered, bio-based wipes perform substantially better in terms of Ionising Radiation and Resource Use - mineral and metals and Resource Use - fossils.

It must be kept in mind that these results exclude the potential impact from the share of product that may end up as littering into the environment. However, this share is not expected to be substantial in regions with a well-developed waste management infrastructure like most EU countries, and no dramatic changes in the overall picture may take place.

### **13.7.7 Interpretation of results related to macro- and micro-plastics generation**

This section discusses the results presented in Section 13.6.3 on the estimated potential generation of macro- and micro-plastics of the assessed wipes scenarios. The latter are compared according to the same criteria applied to the interpretation of the potential environmental impacts (Section 13.7.3), initially focusing on the results from the two-level approach applied as a base case. No sensitivity analyses were assessed for this case study.

Regarding the total loss and release of plastics (macro- and micro-plastics) (Figure 13.13), the two base-case approaches (PLP method and Expanded PLP method) provide almost exact results (<0.01% difference) due to the lack of a recycling share at the End of Life of the product and the potential over-mismanagement of waste in non-EU countries importing plastic waste for recycling. In both cases, no differences are observed among the different scenarios (<0.2%). This is mostly associated to the fact that all scenarios require the same mass to fulfil the functional unit which contribute to the same loss of macro-plastics at the End of Life and limited differences are observed in the supply-chain of the scenarios leading to differences in the loss of micro-plastics related to transportation (e.g. tire abrasion). Note, however, that the applied methods rely on mass-based indicators (which are affected by differences in wipe mass) rather than quantifying the number of items (or plastic fragments/particles) lost or released to the environment. If this was the case, all scenarios would result in the same macro-plastics generation at End of Life when applying both PLP method and Expanded PLP method, since wipes with a lower mass would not imply a reduction in the contribution of product littering and waste mismanagement (which are based on rates that are not affected by the type of feedstock or material). In turn, an identical macro-plastics generation at End of Life would imply having a comparable total loss, these being mostly determined by macro-plastics (see below).

Focusing on the relation between total loss and release (which depends on the parameters of the PLP method), release to ocean represented a lower share (25%) than the release to the terrestrial environment (75%). As shown in Figure 13.15 for the PLP method, the total plastic release to both ocean and the terrestrial environment is dominated by the release of macro-plastics at the End of Life (more than 99%), while micro-plastics released throughout the upstream life cycle have only a minor role. Similar results are also observed for the Expanded PLP method (not shown), despite two

1 additional sources of micro-plastics are considered in the latter (i.e. road markings and  
2 marine coatings). In both cases, this is a consequence of the prevailing mass of macro-  
3 plastics, which is directly related to the mass of wipes required per functional unit  
4 (reference flow), by means of higher loss and release rates compared to micro-plastics.  
5 Therefore, the mass of macro-plastics is at least one order of magnitude higher than the  
6 mass of micro-plastics, which instead depends on the quantity of relevant lifecycle  
7 processes by means of (much) lower loss and release coefficients (depending on the  
8 source).

9 With regard to the origin of released plastics (in terms of life cycle stage and source),  
10 macro-plastics entirely originate from the End of Life stage (Figure 13.16), as they are  
11 only derived from mismanagement of the product as waste. Product littering (assumed  
12 as 2%) represents the 18% of the macro-plastic loss at the End of Life. On micro-plastics  
13 release, the relative contribution of the different sources (e.g. loss of plastic pellets and  
14 tire abrasion) depends on the characteristics of the supply-chain (e.g. feedstock origin,  
15 which affect the contribution from transport across the life cycle). However, within the  
16 limited role of micro-plastics in the overall release to the environment, the most relevant  
17 contribution is generally provided by pellet losses from the Polymer and Article  
18 Production stages, while transport-related sources have a more restricted role.

19 In conclusion, due to the similar product characteristics, supply-chains and modelling  
20 parameters of the different scenarios, no scenario is associated to a better performance  
21 regarding plastic loss and release to the environment.

22

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## 1 **List of abbreviations**

2	ABS	Acrylonitrile butadiene styrene
3	AF	Agri-footprint
4	ALC	Accelerated landfill conditions
5	BOPP	Bi-axially oriented PP
6	BR	Brazil
7	BREF	Best Available Techniques Reference Document
8	C	Carbon
9	C bio	Carbon biogenic
10	CCU	Carbon capture and utilisation
11	CFF	Circular Footprint Formula
12	DE	Germany
13	EC	European Commission
14	ECHA	European Chemicals Agency
15	EF	Environmental Footprint
16	EFTA	European Free Trade Association
17	EI	Ecoinvent
18	EL	Entry level
19	EO	Ethylene Oxide
20	EoL	End of Life
21	EPS	Expanded polystyrene
22	ESR	Electronics Shredder Residue
23	EU	European Union
24	FDCA	Furandicarboxylic Acid
25	FU	Functional Unit
26	GHG	Greenhouse Gas
27	GLO	Global
28	H	Hydrogen
29	HDPE	Polyethylene, high density
30	ILCD	International Reference Life Cycle Data System
31	iLUC	Indirect Land Use Change
32	JRC	Joint Research Centre
33	LC	Life Cycle
34	LCA	Life Cycle Assessment
35	LCI	Life cycle Inventory
36	LCIA	Life Cycle Impact Assessment
37	LDPE	Polyethylene, low density
38	LHV	Lower heating value

1	Lit	Literature
2	LUC	Land Use Change
3	MDI	Methyl Diphenyl Diisocyanate
4	MEA	Monoethanolamine
5	MEG	Mono Ethylene Glycol
6	MI	Material index
7	MSF	Material substitution factor
8	N	Nitrogen
9	O	Oxygen
10	OTR	Oxygen transmission rate
11	PBAT	Polybutylene adipate terephthalate
12	PBS	Polybutylene succinate
13	PC	Polycarbonate
14	P(D)LA	Poly-D-lactic acid
15	PE	Polyethylene
16	PEF	Polyethylene furanoate
17	PEF	Product Environmental Footprint
18	PEFCR	Product Environmental Footprint Category Rules
19	PET	Polyethylene Terephthalate
20	PLA	Polylactic acid
21	P(L)LA	Poly-L-lactic acid
22	PO	Propylene Oxide
23	PP	Polypropylene
24	PS	Polystyrene
25	PTA	Purified Terephthalic Acid
26	PUR	Polyurethane
27	R-ABS	Recycled ABS
28	R-EPS	Recycled EPS
29	R-HDPE	Recycled HDPE
30	R-LDPE	Recycled LDPE
31	R-PC/ABS	Recycled PC/ABS blend
32	R-PET	Recycled PET
33	R-PP	Recycled PP
34	REACH	Registration, Evaluation, Authorisation and Restriction of Chemicals
35	RER	Europe
36	RoW	Rest of the World
37	S	Sulphur
38	SCR	Selective catalytic reduction



Comparative LCA of Alternative Feedstock For Plastics Production – DRAFT FOR  
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1	SNCR	Selective non-catalytic reduction
2	SSP	Solid state polymerisation
3	TPS	Thermoplastic starch
4	TS	Thinkstep
5	TS	Total solids
6	US	United States
7	VS	Volatile solids
8	WEEE	Waste Electric and Electronic Equipment
9		
10		

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Publications Office

doi:XX.XXXX/XXXXXX

ISBN XXX-XX-XX-XXXXX-X