

## JRC TECHNICAL REPORTS

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

*Part 1: the **Plastics LCA** method*

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## **Foreword**

This report represents one of the outcomes of the Administrative Agreement (No. 34854-2017 / DG GROW No. SI2.762599) between the Directorate General for Internal Market, Industry, Entrepreneurship and SMEs (DG GROW) and the Joint Research Centre. It describes a structured and comprehensive methodological framework, referred to as the "*Plastics LCA method*", providing detailed rules to conduct LCA studies of plastic products from different feedstocks, including fossil resources, plastic waste, biomass and CO<sub>2</sub> from gaseous effluents. The method builds upon and conforms to the Product Environmental Footprint (PEF) method, while complementing or specifying further the respective methodological rules wherever required. The overarching aim is to enable as much as possible reproducible, consistent, robust, and verifiable LCA studies of plastic products at the EU level, based on a common and harmonised framework. While the focus is especially on plastic products relying on different feedstocks, the method also applies to products with different biodegradability properties (e.g. compostable plastics), regardless of the feedstock used for production.

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

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Luca Zampori is also sincerely acknowledged for advising on several methodological issues throughout the whole development of the document, for suggestions on how to keep consistency with the provisions, logic and spirit of the PEF method, and for providing useful comments and feedback on introductory sections of this document.

Finally, we are grateful to Alexandre Bouchet and Julien Boucher from *Environmental Action* for providing feedback on the operational description of the *PLP method* reported in Annex I, as well as to Andreas Ciroth from *GreenDelta* for providing useful background information on the approach to estimate plastic litter generation associated with specific life cycle inventory datasets, briefly described in the same annex (Annex I).

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*Simone Nessi* has elaborated an initial draft of this document building upon the provisions from the EU Product Environmental Footprint (PEF) method (Recommendation 2013/179/EU) and the related "PEFCR guidance document" (version 6.3), while developing additional sections on specific methodological or reporting aspects (except those drafted by other authors, as specified below). He has also subsequently revised the document at different stages, and further detailed or complemented it with additional relevant sections. He coordinated the development of sections elaborated by other authors, revised or expanded them as appropriate, and contributed to the related background technical discussion. He also coordinated and contributed to the systematic review of selected LCA studies on plastics described in Annex K, and drafted or revised the related sections. Finally, he was responsible for preparing interim and final versions of the document.

*Taija Sinkko* has taken care of editing the document and improving its layout at different stages. She has also contributed to the systematic review of selected LCA studies on plastics, to drafting part of the related section (Annex K), and to preparing an initial draft

of Annex F (on the relevance of potential indirect effects from fossil-based feedstock supply).

*Claudia Bulgheroni* and *Aikaterini Konti* have contributed to the systematic review of selected LCA studies on plastics, and to editing an early draft of the related section (Annex K).

*Pelayo Garcia-Gutierrez* has elaborated an initial draft of Section 4.4.5, providing modelling rules on the use of CO<sub>2</sub> from gaseous effluents as a feedstock, and of the related background information reported in Annex G.

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*Esther Sanye-Mengual* has drafted Annex I, presenting an overview of approaches currently available to quantify macro- and micro-plastics generation along the product life cycle and detailing the related *Plastic Leak Project* (PLP) method.

*Davide Tonini* has contributed to the definition of rules on the modelling of specific end of life options, by revising Sections 4.4.10.4 to 4.4.10.10 and drafting Section 4.4.10.7 on organic material use-on-land. He has also drafted methodological rules to quantify GHG emissions from indirect Land Use Change (iLUC; Section 4.4.15.3) and developed the related background information on land use changes and on currently available models to estimate GHG emissions from iLUC (reported in Annex J). He also contributed to background discussions on the handling of biogenic carbon emissions and removals (Section 4.4.15.2), on temporary carbon storage and delayed carbon emissions (Section 4.4.15.4), and on the use of CO<sub>2</sub> from gaseous effluents as a feedstock (Section 4.4.5). Finally, he contributed to the initial systematic review of selected LCA studied on plastics (Annex K).

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

## Abstract

The European Strategy for Plastics in a Circular Economy (COM(2018) 28 final) proposes a vision where innovative materials and alternative feedstocks to fossil resources are developed and used for plastics production, where evidence clearly shows that they are more sustainable compared to traditional non-renewable alternatives. Alternative feedstocks to oil and gas include plastic waste, biomass and other bio-based resources, as well as CO<sub>2</sub> from gaseous effluents. The Strategy also urges the identification of those applications where the use of plastics with biodegradable properties (regardless of the feedstock used for production) may provide clear environmental benefits, and the criteria for such applications. It hence calls for work aimed at investigating and better understanding the lifecycle impacts of using alternative feedstocks for plastics production, as well as to develop Life Cycle Assessment (LCA) studies seeking to identify any common condition under which the use of biodegradable or compostable plastics would be beneficial for the environment.

In this context, the Joint Research Centre (JRC) has been requested by DG GROW<sup>1</sup> to elaborate an appropriate LCA-based method to consistently evaluate the potential environmental impacts of plastic products from different feedstocks, while taking into account possible differences in biodegradability properties of the resulting materials. Relevant life cycle-based methods and approaches set out in the existing EU legislation (e.g. the Product Environmental Footprint method<sup>2</sup>) were to be properly taken into account in the development of the method, as well as relevant international and European standards.

Following this request, the JRC has developed a structured and comprehensive methodological framework, referred to as the "*Plastics LCA method*", which is described in this document. The *Plastics LCA method* provides detailed methodological and modelling rules to conduct LCA studies of plastic products based on a common and harmonised framework, with the ultimate aim of enabling as much as possible reproducible, consistent, robust and verifiable studies at the EU level. While the focus is especially on plastic products relying on different feedstocks (i.e. fossil resources, plastic waste, biomass and CO<sub>2</sub>), the method also applies to products with different biodegradability properties (e.g. compostable plastics), regardless of the feedstock used for production.

The *Plastics LCA method* builds upon and conforms to the general structure, logic, and methodological/modelling rules of the latest proposal for updating the Product Environmental Footprint (PEF) method<sup>3</sup>. Complementary or more specific rules have also been included, to cover specific methodological aspects relevant to plastic products (e.g. on the modelling of specific feedstock sources or relevant End of Life options), while keeping full compliance with the overarching rules and principles of the PEF method. Ensuring full consistency between the *Plastics LCA* and the PEF methods has been an overarching aim. However, in case potential conflicts are detected in this document, the rules specified in the PEF method shall prevail.

The development of the method has also taken into account the outcome of an initial systematic review of selected existing LCA studies in the field of plastics, which has highlighted a large variability in applied modelling choices and approaches for a number of relevant methodological aspects, and hence the importance of the need to define common and harmonised rules to ensure increased consistency and reproducibility. The learnings obtained from a number of screening case studies based on a first draft of the

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<sup>1</sup> Directorate General for Internal Market, Industry, Entrepreneurship and SMEs.

<sup>2</sup> Commission Recommendation 2013/179/EU of 9 April 2013 on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations.

<sup>3</sup> Zampori L. and Pant R. (2019) *Suggestions for updating the Product Environmental Footprint (PEF) method*, EUR 29682 EN, Publications Office of the European Union, Luxembourg.

method and readily available data and information has also been taken into account as an essential preliminary input<sup>4</sup>.

Interested stakeholders have been involved at different stages of the method development process, by means of written technical consultations and one dedicated workshop, following a participatory approach. Inputs received from stakeholders have been carefully evaluated and taken into account, as far as appropriate, to produce an advanced and a final draft of the method, from which the final version reported in this document has been derived.

The practical applicability of an advanced draft of the method has been tested and illustrated in ten demonstrative LCA case studies on selected plastic articles, serving the main purpose of supporting the development of the method itself and evaluating its viability and reliability. The description and results of the case studies are reported in a separate supporting document<sup>5</sup>.

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<sup>4</sup> The screening case-studies have been developed to obtain first insights and collect feedbacks on an initial draft of the *Plastics LCA* method. As a consequence, the related report has not been updated further and it is not publicly disclosed. However, the five screening studies have all been refined and developed further into detailed LCA case studies that are available in a separate complementary report (Nessi et al., 2021), as discussed immediately below.

<sup>5</sup> Nessi S., Sinkko T., Bulgheroni C., Garcia-Gutierrez P., Giuntoli J., Konti A., Sanye-Mengual E., Tonini D., Pant R., Marelli L., Ardente F. (2021) Life Cycle Assessment (LCA) of alternative feedstocks for plastics production – Part 2: LCA case studies. Publications Office of the European Union, Luxembourg.

# 1 Introduction

This document provides detailed guidance and rules to conduct Life Cycle Assessment (LCA) studies of plastic products from different feedstocks (including fossil resources, plastic waste, biomass and CO<sub>2</sub> from gaseous effluents), based on a common and harmonised methodological framework at the EU level. The overarching purpose is to enable as much as possible reproducible, consistent, robust and verifiable LCA studies of plastic products. The overall set of rules specified in this document thus defines a structured and comprehensive methodological framework, which is referred to as the “*Plastics LCA method*”.

The *Plastics LCA* method conforms to the general structure and methodological/modelling rules of the Product Environmental Footprint (PEF) method (EC, 2013a), as amended in the update proposal reported in Zampori and Pant (2019). However, where required, the rules of the PEF method are here complemented or detailed further, while keeping full compliance with its overarching rules, principles and logic (see Section 1.6).

The PEF method is a LCA-based method to quantify the potential environmental impacts of products (goods or services) throughout their entire life cycle and across a broad range of relevant environmental impact categories, so as to provide a multi-criteria measure of the lifecycle environmental performance of a product. Since 2013, the PEF method has been recommended by the European Commission as a common harmonised method to measure and communicate the life cycle environmental performance of products at EU level (as per Recommendation 2013/179/EU<sup>6</sup>; EC, 2013b). It has been developed with the aim of establishing a common methodological approach to enable Member States and the private sector to assess, display and benchmark the environmental performance of products or services based on a comprehensive assessment of environmental impacts over the product life cycle, according with one of the core objectives of the “Roadmap to a Resource Efficient Europe” (COM(2011) 571 final)<sup>7</sup>.

The PEF method builds on existing lifecycle approaches and international standards and has the overarching purpose of providing information to enable reducing the environmental impacts of goods and services taking into account (supply-chain) activities throughout their entire life cycle (from extraction and pre-processing of raw materials, through product manufacturing, distribution and use to final waste management of the product at End of Life). A broad range of potential environmental impacts, health effects and resource-related threats are accounted for, rather than focusing only on site-specific impacts or on individual impact categories. Overall, this comprehensive and holistic approach helps preventing unintended shifting of environmental burdens and impacts (“burden shifting”) from one stage of the life cycle to another, from one impact category to another, between environmental/human health impacts and resource efficiency, and/or between countries or regions.

Beyond general, cross-cutting methodological and modelling rules applicable to all product categories, the PEF method also includes technical requirements to develop product category-specific rules (referred to as Product Environmental Footprint Category Rules - PEFCRs) defining a consistent set of additional and/or more specific modelling rules for particular product categories, and to conduct PEF studies in compliance with such specific PEFCRs. Comparisons and comparative assertions among alternative products or product scenarios can be performed only if a PEF study is conducted in compliance with a specific PEFCR. This requirement consistently applies to LCA studies developed according to the *Plastics LCA* method.

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<sup>6</sup> Commission Recommendation of 9 April 2013 on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations.

<sup>7</sup> EC (2011): COM(2011) 571 final. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. Roadmap to a Resource Efficient Europe.

While providing more specific and complementary methodological rules for plastic products and related feedstocks compared to the PEF method, the *Plastics LCA* method does not represent, per se, a PEFCR. This is because it does not fulfil the whole set of requirements for developing PEFCRs reported in Annex A to the PEF method (Zampori and Pant, 2019). For example, it does not address a specific product category with a defined quantifiable function, and does not provide sufficiently specific methodological requirements (e.g. on the inventory data to be applied for specific processes in the product life cycle). However, it addresses a number of relevant methodological and modelling aspects that can be used as a basis for possible future development of PEFCRs for specific categories covering plastic products. On the other hand, specific provisions currently reported in the *Plastics LCA* method may be improved and/or refined further in the future, if more specific experience and knowledge will be gained through the development of PEFCRs for specific product categories.

## 1.1 Context and overall aim

In January 2018, the European Commission adopted the European Strategy for Plastics in a Circular Economy (COM(2018 28 final) (EC, 2018a), proposing a vision where innovative materials and alternative feedstocks to fossil resources are developed and used for plastics production, to reduce dependence on imported fossil fuels and decrease CO<sub>2</sub> emissions. Alternative feedstocks include plastic waste, biomass and other bio-based resources, gaseous effluents (e.g. CO<sub>2</sub>) and should be applied 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 (regardless of the feedstock used for production) may provide clear environmental benefits. Therefore, the Strategy calls for work aimed to investigate and better understand the lifecycle impacts of using alternative feedstocks for plastics production, as well as to develop Life Cycle Assessment (LCA) studies seeking to identify any common conditions under which the use of biodegradable or compostable plastics would be beneficial for the environment.

In this framework, the Joint Research Centre (JRC) has been entrusted by DG GROW<sup>8</sup> with the task of elaborating an appropriate LCA-based method to consistently evaluate the potential environmental impacts of plastic products based on alternative and conventional feedstocks (fossil resources, plastic waste, biomass, CO<sub>2</sub> in gaseous effluents), while taking into account possible differences in biodegradability properties of the resulting materials. Relevant life cycle-based methods and approaches set out in the existing EU legislation (e.g. the Product Environmental Footprint method<sup>9</sup>) should have been properly taken into account in the development of the method, along with methodological learnings from developing the updated Renewable Energy Directive<sup>10</sup>, and relevant European standards for LCA applied to plastic products or related feedstock sources (e.g. EN standards on LCA of bio-based products).

## 1.2 Objective and scope

In line with the overall aims outlined in Section 1.1, the objective of the *Plastics LCA* method presented in this document is to provide common and harmonised methodological rules enabling companies and practitioners to conduct as much as possible consistent, reproducible, robust and verifiable LCA studies of plastic products from different feedstock sources at the EU level. While focusing more specifically on the use of different feedstocks, the method also addresses plastic products with different biodegradability properties<sup>11</sup>, regardless of the feedstock used for production.

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<sup>8</sup> Directorate General for Internal Market, Industry, Entrepreneurship and SMEs.

<sup>9</sup> Recommendation 2013/179/EU.

<sup>10</sup> Directive (EU) 2018/2001 of the European Parliament and of the Council of 11 December 2018 on the promotion of the use of energy from renewable sources (recast).

<sup>11</sup> See the list of definitions for a more detailed description of how biodegradable plastics and biodegradability are intended for the purpose of this method.

The *Plastics LCA* method intends to especially address “micro-level” LCA studies on specific products with a quantifiable function, adopting a value-chain perspective. It does not address macro-level assessments considering the effects of strategies or broader policy decisions/initiatives implying relevant changes at society/industry level<sup>12</sup> (e.g. due to large-scale variations in the type of feedstock or material used to manufacture a given plastic product at the EU level or at any other large geographical scale).

Conforming to the PEF method, the *Plastics LCA* method especially applies to commercially available products (i.e. products available on the market) manufactured in industrial-scale facilities (and only in this case the LCA results can be used for external communication; see also Section 1.7). However, the methodological rules specified in this document may be equally applied also to non-commercially available products (regardless of the development level of the underlying production/manufacturing technology)<sup>13</sup>, provided that the study relies on (measured) company-specific data from real facilities for the product manufacturing process, and that the LCA results are used only for company-internal applications (i.e. they are not communicated externally).

While the *Plastics LCA* method can be indifferently applied to any kind of plastic product and related feedstocks, comparisons and/or comparative assertions among different products or product scenarios providing equivalent functions can be performed only if the results are based on the same Product Category Rules (PEFCRs; see Section 2.6). One of the aim of PEFCRs is indeed to fix a consistent and specific set of methodological and modelling rules for products belonging to a same common category, to enable comparisons and comparative assertions wherever this is feasible, relevant and appropriate (see Table A-1 in Annex A to the PEF method for details; Zampori and Pant, 2019). This is achieved, for instance, by aiming to an equivalent accuracy and comparable data quality for all products or product scenarios that may be compared within the specific category.

### 1.3 Development process

The *Plastics LCA* method has been developed following a participatory approach, involving interested stakeholders at different stages of the development process. This has taken place through both dedicated technical consultations (written and/or oral) and calls for data and information, as described below.

A first draft of the method has been initially elaborated, building upon methodological rules specified in the PEF method (EC, 2013a) and in the related PEFCR guidance document (version 6.3; EC, 2018b)<sup>14</sup>, as well as taking into account the outcome of a systematic review of selected existing LCA studies in the field of plastics (summarised in Annex K). In particular, the latter highlighted a large variability in applied modelling choices and approaches for a number of relevant methodological aspects, and hence the importance of the need to define common and harmonised rules to ensure increased consistency and reproducibility. The applicability and reliability of the draft method have also been tested in five screening case studies based on readily available data and information, and accounting for the inputs received through a specific call for data and information open to all interested stakeholders, held in summer 2018. The draft method and the outcome of the screening case studies have then been submitted to a technical stakeholder consultation to collect useful feedback, comments and suggestions for improvement. The consultation consisted of both a written consultation process (21 November – 19 December 2018), and a stakeholder workshop (held in Brussels on 29-30 November 2018).

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<sup>12</sup> For macro-level assessments that consider society/industry level changes, dedicated guidance is available in the ILCD Handbook - Detailed guidance (under Situation B; EC-JRC, 2010). However, such guidance is not specific for plastics and plastic products. Elaboration of dedicated guidance might be subject of later work, while it is noted that far less practical experience actually exists with such type of assessments.

<sup>13</sup> I.e. regardless of whether they are based on an (established) industrial/commercial-scale technology or a new/emerging technology.

<sup>14</sup> EC (2018b) Guidance for the development of Product Environmental Footprint Category Rules (PEFCRs), version 6.3, May 2018.

Following the consultation, a revised version of the *Plastics LCA* method has been developed, taking into account the inputs received from stakeholders, the lessons learnt from the screening case studies, and the proposed suggestions for updating the Product Environmental Footprint method made available by Zampori and Pant (2019). Based on the revised version of the method, a number of demonstrative LCA case studies on ten selected plastic products have been carried out, with the aim to test and illustrate the practical applicability of the method, and to evaluate its reliability. To ensure proper and transparent development of the case studies, a second call for data and information was run in spring 2019, allowing interested stakeholders to provide comments on selected scenarios and functional units, as well as relevant data and technical information to be used in the studies. Stakeholders were invited again to provide feedback, comments and suggestions on both the updated *Plastics LCA* method and on the illustrative LCA case studies, in the framework of a written stakeholder consultation that took place from 3 to 30 June 2020.

Inputs received from this second consultation were analysed and taken into account, as far as appropriate, to elaborate a final draft of the *Plastics LCA* method. This has been submitted to a review by external experts to evaluate whether stakeholder comments had been implemented consistently with the rules in the latest version of the PEF method (Zampori and Pant, 2019), and to ensure full compliance with the latter. Based on the outcome of the review, a final version of the method was finally developed, as reported in this document.

## 1.4 Target audience

This document is primarily aimed at technical experts who need to develop LCA studies of plastic products in compliance with the *Plastics LCA* and the PEF methods. Example of users include engineers and environmental managers in companies and other institutions.

## 1.5 Relationship to other methods and standards

The methodological and modelling provisions specified in the *Plastics LCA* method have been developed based on the requirements and recommendations of the Product Environmental Footprint (PEF) method, as amended in the update proposal by Zampori and Pant (2019). This has been the main reference, and ensuring full compliance with it has been an overarching aim. Where required, provisions from the PEF method have been complemented or specified further (see Section 1.6), to be made more specific and tailored to plastic products, which are in the scope of this document.

The requirements specified in the PEF method have been developed taking into account the requirements and recommendations of similar, widely recognised product environmental accounting methods and guidance documents, including also ISO 14040 and 14044 standards on Life Cycle Assessment (ISO 14040:2006; ISO 14044:2006). The full list is available in Section 1.4 of the PEF method (Zampori and Pant, 2019), where the reader is referred to for further detail.

Additionally, the following standards and legislation have been specifically considered for the development of the *Plastics LCA* method<sup>15</sup>:

— EN standards:

- EN 16760:2015 Bio-based products – Life Cycle Assessment;
- CEN TR 16957:2016 Bio-based products – Guidelines for Life Cycle Inventory (LCI) for the End-of-life phase;

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<sup>15</sup> Note that this does not imply that a LCA study developed based on the *Plastics LCA* method will automatically comply with such standards and legislation, nor with ISO 14040 and 14044 LCA standards mentioned above.

- EN 13432:2000 Packaging - Requirements for packaging recoverable through composting and biodegradation - Test scheme and evaluation criteria for the final acceptance of packaging;
  - EN 14995:2006 Plastics - Evaluation of compostability - Test scheme and specifications;
  - EN 17033:2018 Plastics - Biodegradable mulch films for use in agriculture and horticulture - Requirements and test methods;
- ISO standards:
- ISO 14855-1:2012 Determination of the ultimate aerobic biodegradability of plastic materials under controlled composting conditions – Method by analysis of evolved carbon dioxide – Part 1: General method;
  - ISO 15985:2014 Plastics – Determination of the ultimate anaerobic biodegradation under high-solids anaerobic digestion conditions – Method by analysis of released biogas;
  - ISO 17556:2012 Plastics – Determination of the ultimate aerobic biodegradability of plastic materials in soil by measuring the oxygen demand in a respirometer or the amount of carbon dioxide evolved;
- Directive (EU) 2018/2001 of the European Parliament and of the Council of 11 December 2018 on the promotion of the use of energy from renewable sources (recast) (EC, 2018c);
- Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC (EC, 2009);
- Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives (consolidated text) (EC, 2008);

Whereas most existing methods may provide several alternatives for a given methodological decision point, the intention of the *Plastics LCA* method is (wherever feasible, and in line with the approach adopted in the PEF method) to identify a single requirement or recommendation for each decision point, or to provide additional guidance that will support more consistent, robust and reproducible LCA studies for plastic products. Thus, reproducibility is given priority over flexibility. This should however not be taken to restrict or limit the necessity for relevant and appropriate sensitivity analysis on specific modelling assumptions (i.e. not on the modelling rules specified in this method), such as more uncertain assumption that considerably affect the results, to test the robustness of results in a LCA study.

## **1.6 Main additions and differences compared to the PEF method**

Compared to the Product Environmental Footprint (PEF) method, specific rules have been added or specified (and possibly discussed) further in this document, to better and/or properly cover relevant methodological or modelling issues for plastic products and the related variety of feedstocks. On a general level, these additions and further specifications include:

- Inclusion of additional provisions for a proper description and reporting of the product(s) in scope, of its technical characteristics and of its relevant function(s);
- Further specification of rules related to the definition of the functional unit (particularly also including the “where” and “for whom” aspects), the calculation of the reference flow, and inclusion of specific examples for plastic products;
- Description of the main potential “indirect effects” associated with plastics value chains, and clarification on the exclusion of such effects from the system boundary of *Plastics LCA* studies;

- Definition of the specific type of “additional environmental information” that shall/should/may be included in a *Plastics LCA* study (and of the related calculation method, where relevant);
- Further specification of processes and activities to be included under the different life cycle stages<sup>16</sup> in the case of plastic products, differentiating, when relevant, by type of feedstock used for production (e.g. for the stage of Raw Material Acquisition and Pre-processing);
- Definition of more specific rules to handle and model losses/waste of products, packaging and packaged product occurring during the Distribution and Use stages, especially when food packaging is the product in scope<sup>17</sup>;
- Definition of general or more specific rules to model the supply<sup>18</sup> of relevant feedstocks for plastic products, i.e. fossil-based resources, plastic waste, primary biomass, bio-based waste/by-products, and CO<sub>2</sub> captured from gaseous effluents;
- Further specification of a number of modelling rules related to agricultural production, or restriction of these (e.g. to calculate NO<sub>3</sub>-N emissions from leaching after fertilisers application);
- Inclusion of general recommendations on the handling of plastic products relying on new, emerging, or maturing technologies or processes;
- Inclusion of general provisions to define appropriate average End of Life scenarios for the product in scope (including non-commercially available products);
- Further specification of modelling rules related to the handling of pre-consumer waste/scrap under the Circular Footprint Formula;
- Inclusion of general and specific rules to model relevant End of Life options for plastic products, including mechanical recycling, biological treatments (industrial composting and anaerobic digestion), on-land application of organic material from biological treatments, in-situ biodegradation (on/into the soil), incineration, landfilling, and release of plastic products in the environment;
- Further specification/description of rules to quantify GHG emissions from direct Land Use Change and changes in soil carbon stocks;
- Provision of calculation rules to quantify GHG emissions from indirect Land Use Change (iLUC) for the purpose of providing “additional environmental information”;
- Recommendation of a preliminary method to quantify macro- and micro-plastics generation and release (including product litter) throughout the product life cycle, for the purpose of providing “additional environmental information”.

Other than the additions and further specifications listed above, the examples provided throughout the PEF method have been as far as possible adapted to better reflect the specific focus of this document on plastic products. Where relevant and needed, additional examples have also been provided, as well as additional or more specific background information, description or discussion on specific methodological or modelling issues, to facilitate their understanding by users of this method. On the other hand, some sections or set of rules reported in the PEF method, and considered less relevant for the specific scope of this document, have not been explicitly reported, to limit its overall length and to facilitate reading<sup>19</sup>. While excluded, such provisions are clearly mentioned

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<sup>16</sup> Raw Material Acquisition and Pre-processing, Manufacturing, Distribution stage, Use stage, and End of Life.

<sup>17</sup> Note that the development of a PEFCR on “Flexible packaging” is currently (April 2021) ongoing. In case of contrasts, the requirements of such PEFCR shall prevail over those in this *Plastics LCA* Method, once the final PEFCR will be approved.

<sup>18</sup> In most cases corresponding to the stage of Raw Material Acquisition and Pre-processing.

<sup>19</sup> Sections of the PEF method not included in this document are those providing detailed modelling requirements for electricity use (Section 4.4.2 of the PEF method), detailed rules to apply the sampling procedure (Sections 4.4.6.1-4.4.6.4 of the PEF method), provisions on the handling of multi-functional

and referenced throughout the document, and shall be applied wherever relevant to conduct a *Plastics LCA* study for the product in scope. Finally, note that the structure of this report has been partially modified compared to the PEF method, to better reflect the specific purpose of this document. Annex A provides a detailed overview of additional and more specific requirements, recommendations, information and examples provided in this document compared to the PEF method, as well as a list of the changes made to its overall structure.

## 1.7 Examples for potential applications

Results of *Plastics LCA* studies are the basis for the provision of information on the lifecycle environmental performance of plastic products. *Plastics LCA* studies may be used in a diverse number of potential fields of application. These include:

- In-house applications (for both commercially and non-commercially available products on the market; results are not communicated externally):
  - Optimisation of processes along the life cycle of a product,
  - Support to environmental management,
  - Identification of environmental hotspots,
  - Support for product design minimising environmental impacts along the life cycle,
  - Environmental performance improvement and tracking,
- External applications: (e.g. business to business (B2B), business to consumer (B2C); limited to commercially available products; results can be communicated externally):
  - Responding to customers and consumers demands,
  - Marketing,
  - Co-operation along supply chains to optimise the product along the life cycle,
  - Participation in 3<sup>rd</sup> party schemes related to environmental claims or giving visibility to products that calculate and communicate their life cycle environmental performance.

The *Plastics LCA* method can be used as a basis for the development of PEFCRs covering plastic products. When a LCA study is conducted in compliance with an existing PEFCR for the product in scope, potential applications in addition to the ones listed above, are:

- Comparisons and comparative assertions (i.e. claims of overall superiority or equivalence of the environmental performance of one product compared to another (based on ISO 14040:2006)), wherever allowed by the relevant PEFCR;
- Comparison and comparative assertions against the benchmark of the product category followed by a grading of other products according to their performance versus the benchmark;
- Identification of significant environmental impacts common to a product group;
- Reputational schemes giving visibility to products that calculate their life cycle environmental performance;
- Green procurement (public and corporate).

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processes during animal husbandry (Sections 4.5.1.1-4.5.1.8 of the PEF method), as well as specific requirements to develop PEFCRs and the related PEFCR template (Annexes A and B of the PEF method).

## **1.8 Available documents**

This report presents the final version of the *Plastics LCA* method, including the entire set of related methodological/modelling rules. It also includes the description and the main results and conclusions of the systematic review of selected existing LCA studies in the field of plastics, initially conducted to inform and support the development of a first draft of the *Plastics LCA* method. This information is available in Annex K.

The description and results of the demonstrative LCA case studies on ten selected plastic products are also available, and reported in a separate document (Nessi et al., 2021). As mentioned earlier, the case studies had the purpose to support the development of the *Plastics LCA* method, to test and demonstrate its practical applicability, and to evaluate its reliability. The procedure initially applied to identify relevant plastic products to be considered in the case studies, which were ultimately selected taking into account stakeholders' feedback received at a later stage, is also reported as an annex to the same report.

An additional interim working document was also produced during the development of the *Plastics LCA* method, including a description and the main outcomes of five screening case studies conducted to test the applicability and reliability of a first draft of the method itself. The document had the main purpose to inform a first stakeholder consultation and collect feedbacks on such a first draft. As a consequence, it has not been updated further and it is not publicly disclosed. However, the screening studies have all been refined, amended and developed further into detailed LCA case studies used to test, demonstrate and evaluate applicability of the final *Plastics LCA* method. As such, they are available in the complementary report mentioned above (Nessi et al., 2021).

## **2 General considerations and requirements for Plastics LCA studies**

### **2.1 General requirements for *Plastics LCA* studies**

To perform a LCA study conforming to this method, the two following general requirements shall be met:

1. The Bill of Materials (BoM) shall be specific of the product in scope; and
2. The modelling of the manufacturing processes shall be based on company-specific data (e.g. energy needed for the assembly of the materials/components of the product in scope).

For companies producing more than one product, the activity data used (including the BoM) shall be specific to the product in scope of the study. See Section 4.6.1 for further details.

### **2.2 How to use this method**

This method provides methodological and modelling rules to conduct as far as possible consistent, reproducible, robust, and verifiable LCA studies of plastic products. Rules are presented in a sequential manner, in the order of the methodological phases and steps that shall be completed when conducting a LCA study.

Each section of the document corresponds to a specific phase or step, and normally begins with a general description of the relevant phase/step, and/or providing general necessary considerations on the addressed methodological issue. This is followed by the methodological and modelling requirements and/or recommendations that apply to the specific phase, step or aspect. Supporting and illustrative examples are also provided where appropriate. In few cases, "Tips" describing non-mandatory but recommended best practices are reported.

### **2.3 Terminology used: shall, should and may**

This method uses precise terminology to indicate the requirements, the recommendations and the options that a practitioner may chose when developing a compliant LCA.

The term "**shall**" is used to indicate what is required in order for a LCA study to be in conformance with the *Plastics LCA* method.

The term "**should**" is used to indicate a recommendation rather than a requirement. Any deviation from a "should" requirement has to be justified by the user of the *Plastics LCA* method, and made transparent.

The term "**may**" is used to indicate an option that is permissible. Whenever options are available, the LCA study report shall include adequate argumentation to justify the chosen option.

Where only specific deviations are eligible or specific conditions apply (for the "should" and "may" cases), this is made explicit in the document.

### **2.4 Principles for *Plastics LCA* studies**

To produce reliable, reproducible and verifiable *Plastics LCA* studies, a core suite of analytical principles shall be strictly adhered to. They provide overarching guidance in the application of this method, and shall be considered with respect to each phase of *Plastics LCA* studies, from the definition of goal and scope, through data collection, impact assessment, reporting and verification of study outcomes (see Section 2.5).

The following principles shall be observed by users of this method in conducting a *Plastics LCA* study:

1. Relevance

All methods used and data collected for the purpose of the LCA study shall be as relevant to the study as possible.

2. Completeness

LCA studies shall include all environmentally relevant material/energy flows and other environmental burdens as required for adherence to the defined system boundary, the data requirements, and the impact assessment methods applied.

3. Consistency

Strict conformity to this method shall be observed in all steps of the LCA study so as to ensure internal consistency and comparability with similar analyses.

4. Accuracy/Quality

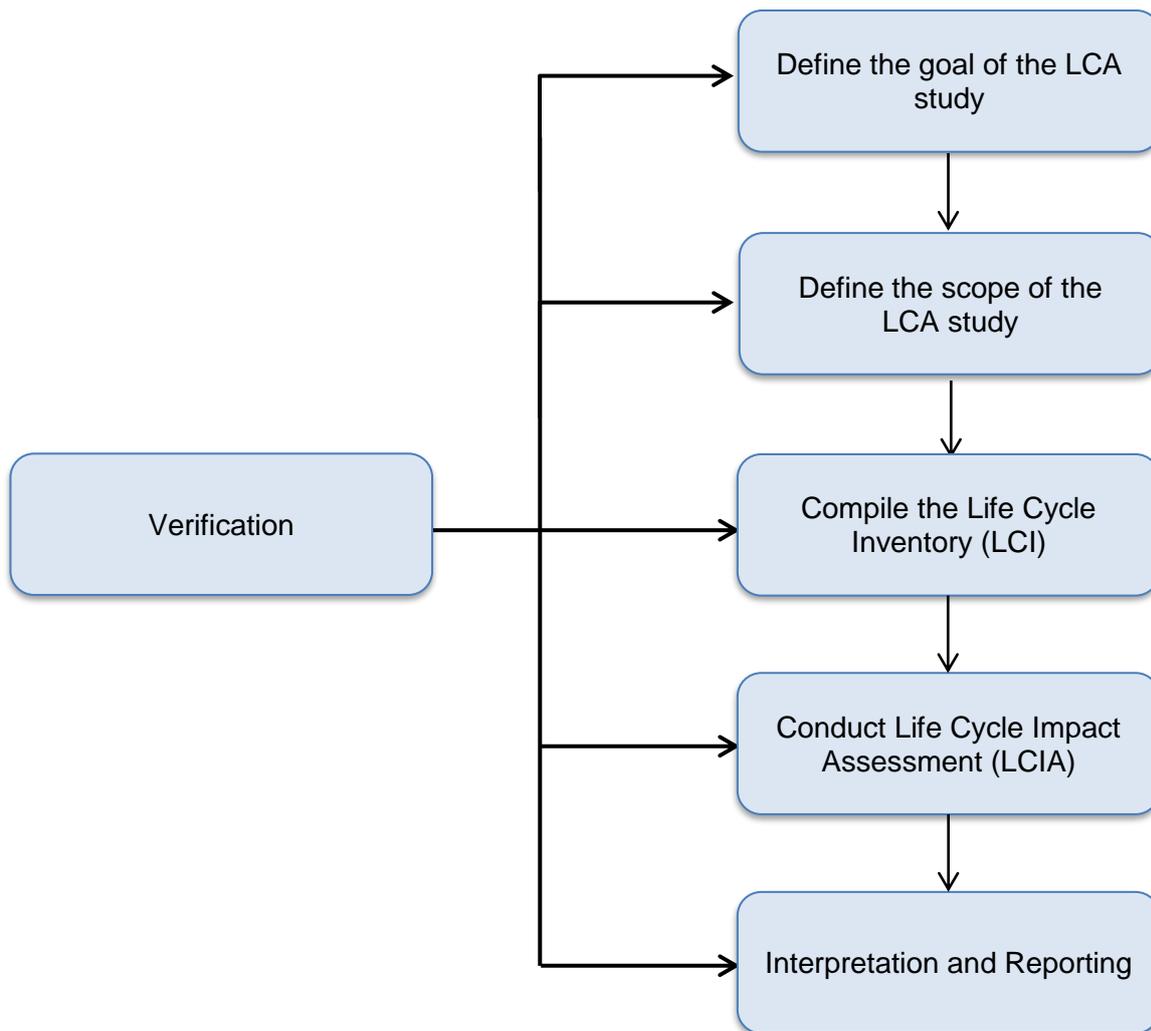
All reasonable efforts shall be taken to increase quality of the inventory data applied for modelling of a product life cycle (i.e. to increase their technological, time-related (age) and geographical representativeness, and reduced uncertainty) and in the reporting of results.

5. Transparency

LCA information shall be disclosed in such a way as to provide intended users with the necessary basis for decision-making, and for stakeholders to assess its robustness and reliability. Confidentiality concerns and protection of proprietary data shall be considered and can be solved, for instance, via review under confidentiality agreements.

## **2.5 Phases of a *Plastics LCA* study**

A number of phases shall be completed to carry out a LCA study in line with this method, conforming to the overall structure of PEF studies with only minor and non-substantial deviations (Figure 1): Goal Definition, Scope Definition, Life Cycle Inventory, Life Cycle Impact Assessment, Interpretation of LCA results and LCA Reporting.



**Figure 1.** Phases of a LCA study in line with the *Plastics LCA* method (adapted from the Product Environmental Footprint (PEF) method; Zampori and Pant, 2019).

In the goal definition phase, the aims of the study are defined, including the intended application, the reasons for carrying out the study and the intended audience (Section 3.1).

Main methodological choices are made in the scope definition phase, derived from and in line with the goal phase. These include, for example, the definition of the functional unit, the identification of the system boundary, the selection of additional environmental and technical information considered in the study, as well as the main assumptions and limitations<sup>20</sup>.

The life cycle inventory (LCI) phase involves data collection and the calculation procedure for the quantification of relevant inputs and outputs of the studied system(s). Inputs and outputs concern energy, raw materials and other physical inputs, products and co-products, any generated waste, as well as emissions to air/water/soil. Data collected refer to both foreground processes<sup>21</sup> and background processes<sup>22</sup>. Data are put in

<sup>20</sup> Note that a comprehensive list of assumptions and limitations can normally be made only after having completed the next life cycle inventory phase, so that any gaps or limitations in terms of data and/or methods are known for the studied systems.

<sup>21</sup> Foreground processes are those processes in the product life cycle for which direct access to information is available to the practitioner. These include, for instance, processes run at the producer's site, and other processes operated by the producer or its contractors (e.g. transport of goods, head-office services, etc.).

relationship to the intended input or output of specific process units, and then to the functional unit of the study. The LCI is an iterative process: as data are collected and more is learned about the system, new data requirements or limitations may be identified that require a change in the data collection procedures so that the goal of the study will still be met.

In the impact assessment phase, LCI results are associated to environmental impact categories and indicators. This is done through LCIA methods, which first classify emissions into impact categories and then characterize (i.e. convert) them to common units (e.g. CO<sub>2</sub> and CH<sub>4</sub> emissions are both expressed in CO<sub>2</sub>-equivalent emissions by means of their global warming potential<sup>23</sup>). Examples of impact categories are climate change, acidification or resource use.

Finally, in the interpretation phase, results from both LCI and LCIA are interpreted in accordance to the stated goal and scope. In this phase, most relevant impact categories, life cycle stages, processes and elementary flows are identified. Conclusions and recommendations can be drawn, based on the analytical results. All relevant items of the study phases are finally documented in a LCA study report.

Verification is mandatory whenever the LCA study, or part of the information therein, is used for any type of external communication (i.e. non-confidential communication to any interested party other than the commissioner or the user of the LCA study).

## **2.6 Product Environmental Footprint Category Rules (PEFCRs) and relation with the *Plastics LCA* method**

The primary objective of a PEFCR is to fix a consistent and specific set of rules to calculate the relevant environmental information of products belonging to the particular product category in scope. Examples of product categories relevant for plastic products include, for instance, beverage bottles, flexible packaging, agricultural and horticultural mulching films, and thermal insulation for buildings. An important objective of PEFCRs is to focus on what matters most for a specific product category to make PEF studies easier, faster and less costly.

An equally important objective is to enable comparisons and comparative assertions in all cases where this is feasible, relevant and appropriate. Comparisons and comparative assertions are allowed only if PEF studies are conducted in compliance with a PEFCR. A PEF study shall be conducted in compliance with a PEFCR, if a PEFCR is available for the product in scope. The same requirements identically apply also to LCA studies developed following the *Plastics LCA* method, which may be seen as an intermediate, “umbrella-type” guidance document between the PEF Method and more specific PEFCRs on defined categories of plastic products (or wider product categories that include plastic products).

Requirements for the development of PEFCRs are specified in Annex A to the updated PEF method (Zampori and Pant, 2019). A PEFCR may further specify requirements made in the PEF or *Plastics LCA* methods, add new requirements where the PEF or *Plastics LCA* methods leaves more than one choice, or derive requirements on products made from other materials than plastic and covered under the same PEFCR (e.g. flexible packaging). The objective is to ensure that PEFCRs are developed according to the PEF method, and that they provide the specifications needed to achieve the comparability, increased reproducibility, consistency, relevance, focus and efficiency of PEF/*Plastics LCA* studies.

Rules similar to PEFCRs exist in standards for other types of life cycle based product claims, such as ISO 14025:2006 (type III environmental declarations). Those were found

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<sup>22</sup> Background processes are those processes in the product life cycle for which no direct access to information is possible to the practitioner. For example, most of the upstream life-cycle processes from product manufacturing, and generally all processes further downstream, may be part of the background processes.

<sup>23</sup> Global Warming Potentials (GWPs) considered in the *Plastics LCA* method (and referred to throughout this document) are those provided in the 5<sup>th</sup> Assessment Report of the IPCC for a time horizon of 100 years and including carbon feedback mechanisms (IPCC, 2013).

to be not sufficiently reproducible and sometimes not sufficiently balanced to achieve the required objectives of PEF studies based on category rules. PEFCRs were named differently to prevent confusion with other similar rules and to uniquely identify rules under the PEF method.

PEFCRs should, to the extent possible, and recognising the different application contexts, be in conformity with existing relevant international Product Category Rules (PCR). If other PCRs are available from other schemes, these are to be listed and evaluated. They may be used as a basis for developing a PEFCR, in line with the requirements provided in Annex A to the updated PEF method (Zampori and Pant, 2019).

## **2.7 Outlook**

It is anticipated that the *Plastics LCA* Method may need to be updated and/or complemented once the PEF Method will be finalised for use after the ongoing transition phase of the Environmental Footprint Initiative (2019-2021).

### 3 Defining the goal(s) and scope of the LCA study

#### 3.1 Goal definition

Goal definition is the first step of a LCA study, and sets the overall context for the study itself. The purpose of clearly defining goals is to ensure that the analytical aims, methods, results and intended applications are optimally aligned, and that a shared vision is in place to guide participants in the study. Therefore, it is important to take the time to carefully consider and articulate goals in order to ensure the success of the LCA study. On the other hand, the decision to use this method (which builds upon the PEF method) implies that some aspects of the goal definition will be decided a priori, due to the specific requirements provided by the PEF method itself.

In defining goals, it is important to identify the intended applications and the degree of analytical depth and rigour of the study. This should also be reflected in the defined study limitations (Scope definition phase, Section 3.2). In particular, the goal definition of the LCA study shall address the following aspects, which are further exemplified in Table 1:

- Intended application(s);
- Reasons for carrying out the study and decision context;
- Target audience;
- Commissioner of the study;
- Identity of the verifier or of the members of the verification team (in this case the lead verifier shall be specified).

**Table 1.** Example of goal definition (LCA of disposable shopping bags).

Aspect	Description
Intended application(s):	Evaluate the potential environmental impacts of a new type of biodegradable shopping bag made of partly bio-based polymers
Reasons for carrying out the study and decision context:	Provide information on the lifecycle environmental performance of the product to customers
Target audience:	External technical audience, business-to-business
Commissioner of the study:	Bags&Bags limited
Verifier:	Independent external verifier (Dr. Name Surname, affiliation)

#### 3.2 Scope Definition

Beyond the initial general information on the product system(s) to be analysed given in the goal definition, additional details need to be provided in the scope definition phase of the LCA study. Especially, when the goal of the study is of a less specified nature, the system(s) to be analysed (e.g. the specific packaging option(s)) still need to be identified and specified in detail. This shall be done in the scope phase of the LCA study. The need for such a better specification in the scope definition is always found when the goal relates to e.g. "generic", "average", "concept" or other insufficiently defined product characteristics that need interpretation.

The scope of the LCA study thus describes in detail the system to be evaluated and the related technical specifications.

The scope definition shall be in line with the defined goals of the study and shall include the following elements (see subsequent sections for a more detailed description):

- Description and characteristics of the product in scope and of the respective life cycle (supply chain, Use Stage and End of Life);
- Functional unit and reference flow;
- System boundary;
- Impact categories;
- Additional information included in the study;
- Assumptions/Limitations.

### 3.2.1 Description and characteristics of the product(s)

A general description of the product(s) in scope and of its relevant function(s) shall be included in the LCA study report. Moreover, the technical properties and characteristics reported in Table 2 shall be specified for the product(s), along with the respective source (where relevant). This ensures transparency and enables reproducibility and comparability of the study. Since the reported information is essential for a proper development of the life cycle inventory, it shall be made accessible to the practitioner of the study, whenever available. However, part of such information may be considered confidential, and hence not disclosed in the LCA study report (as specified in Table 2).

The Bill of Materials (BoM) of the specific product(s) (including packaging), i.e. its breakdown by mass, shall be provided, differentiating between the actual product and its packaging (if any). The BoM of the actual product(s) should be further differentiated by each part or component, if data are available. The total mass of the product(s) shall also be reported.

A general description of the product(s) life cycle shall also be provided, including the main processes and activities involved in the respective supply chain, Use Stage and End of Life. Relevant details related to logistics and transport throughout the life cycle should also be reported.

**Table 2.** Technical properties of the product(s) in scope that shall be specified in the study. This information should be reported separately for each of the (main) product/packaging components<sup>24</sup> (e.g. including printed labels, lids and caps), if data is available. For composite materials, information should be provided for each single material.

Property / Parameter	Unit (recommended)
<b><i>Main constituents and respective feedstock</i></b>	
Main polymer	g/kg <sup>(3)</sup>
Co-monomer/Co-polymer 1	
- <i>Co-monomer 1</i>	
- <i>Co-monomer 2</i>	
- <i>Co-monomer n</i>	
Co-monomer/Co-polymer n	
- <i>Co-monomer 1</i>	
- <i>Co-monomer n</i>	

<sup>24</sup> Minor components (i.e. lower than 1% by mass) may be excluded (e.g. printed labels).

<b>Property / Parameter</b>	<b>Unit (recommended)</b>
Additive 1 (type of additive, e.g. plasticiser, filler, flame retardant, stabiliser) <sup>(1, 2)</sup>	
Additive 2 (type of additive, e.g. plasticiser, filler, flame retardant, stabiliser) <sup>(1, 2)</sup>	
Additive n (type of additive, e.g. plasticiser, filler, flame retardant, stabiliser) <sup>(1, 2)</sup>	
<b>Biodegradability properties <sup>(4)</sup></b>	
Compostability <sup>(5)</sup>	(Y/N) specifying the standard of reference <sup>(7)</sup> and the corresponding biodegradation rate (%)
Anaerobic treatability <sup>(6)</sup>	
Biodegradability -terrestrial-	(Y/N) specifying any standard of reference type of biodegradability (ready or inherent) and the corresponding biodegradation rate (%)
Biodegradability -marine-	
<b>Other properties</b>	
Lifetime (expected or estimated)	days / years
Reusability	Multi-use / Single-use
Density	kg/m <sup>3</sup>
Energy content -as lower heating value (LHV)-	MJ/kg <sup>(3)</sup>
<b>Chemical composition <sup>(8)</sup></b>	
Water content	% <sup>(3)</sup>
Volatile Solids (VS)	
Ash	
Carbon -fossil- (C <sub>fossil</sub> )	g/kg <sup>(3)</sup>
Carbon -biogenic- (C <sub>bio</sub> )	
Hydrogen (H)	
Oxygen (O)	
Nitrogen (N)	
Phosphorus (P)	
Potassium (K)	
Sulphur (S)	
Chlorine (Cl)	
Fluorine (F)	
Arsenic (As)	
Cadmium (Cd)	
Cobalt (Co)	
Chromium (Cr)	
Copper (Cu)	
Lead (Pb)	

Property / Parameter	Unit (recommended)
Manganese (Mg)	
Mercury (Hg)	
Nickel (Ni)	
Zinc (Zn)	
Other elements (e.g. Se and Mo)	

- (<sup>1</sup>) While the type of additive used shall always be reported (e.g. plasticiser, filler, flame retardant, stabiliser), the specific substance used and its quantity should also be specified whenever this cannot be disclosed for confidentiality reasons.
- (<sup>2</sup>) Additives used during product manufacturing (i.e. polymer conversion) processes shall also be considered (e.g. antioxidants), although these may not be explicitly included in the product formulation.
- (<sup>3</sup>) All contents and composition values are to be expressed in relation to the overall (wet) mass of the product or component.
- (<sup>4</sup>) Information on compostability, anaerobic treatability, and biodegradability in the open environment are only required where such properties are claimed for the product in scope. For non-biodegradable or poorly biodegradable polymers, a value equal to "0" can be reported without reference to any standard.
- (<sup>5</sup>) Including, but not limited to, inherent and ultimate aerobic biodegradability under controlled composting conditions.
- (<sup>6</sup>) Including, but not limited to, inherent and ultimate anaerobic biodegradability under (high-solids) anaerobic digestion conditions.
- (<sup>7</sup>) For instance EN 13432:2000 or EN 14995:2006.
- (<sup>8</sup>) While the chemical composition may be difficult to know without a specific analysis, it is essential to develop a proper life cycle inventory, especially (but not only) to calculate emissions from the End of Life stage, and if uncommon or complex materials are used in the product. The chemical composition should hence be provided to the practitioner, and reported in the LCA study report whenever it cannot be disclosed for confidentiality reasons.

### 3.2.2 Function of the product, functional/declared unit and reference flow

The functional unit (FU) is the quantified performance of a product system, to be used as a reference to calculate the Life Cycle inventory of a product and its potential environmental impacts. The functional unit quantifies the selected relevant product function(s), their level of quality, and the duration over which they are to be provided. For instance, the FU of a polymer coating could be described as *"providing protection of 1 m<sup>2</sup> of substrate for 20 years, with a minimum 98% opacity, in Europe"* (further examples are provided in Section 3.2.2.4 for selected plastic products). Meaningful comparisons under a specific PEFCR can only be made when products can fulfil the same function.

The reference flow is the amount of product needed to provide the defined functional unit. All input and output flows of processes and activities included in the analysis quantitatively relate to it. For instance, the reference flow for a polymer coating could be the volume or mass of coating needed to fulfil the functional unit specified above.

Users of this method shall specify the function(s) of the analysed product(s) considered as relevant for the LCA study, and to define the corresponding functional unit and reference flow. Specific provisions and examples are reported in the following subsections (3.2.2.1 – 3.2.2.4).

#### 3.2.2.1 Function(s) of the product

The function(s) of the product(s) in scope that are selected as relevant for the LCA study, and are hence retained in the functional unit, shall be clearly specified. A product may perform more than one function at the same time, and the one(s) selected as relevant for the study depend on its goal and scope. For instance, a polymer coating can provide both surface protection and colouring/decoration. If the LCA study focuses on interior coatings, surface protection may be considered unnecessary (or secondary) for the specific goal and scope, while colouring/decoration could be selected as the relevant

function. Similarly, an electronic device may be considered to also provide heating to the surrounding environment where it is used, beyond performing the primary function it is intended for.

Any omission of the functions of the product in the definition of the functional unit shall be explained and documented. Any relevant additional or secondary product function that is not taken into account in the functional unit should be listed, and its exclusion shall be justified when this has an influence on the resulting reference flow.

If additional or secondary functions are included in the study, and different product alternatives are compared under a specific PEFCR, full functional equivalence shall be ensured among any of such alternatives. For instance, if an electronic device also provides substantial heating to the surrounding environment, this additional function may be included in the FU, and shall be equally provided by any alternative devices possibly compared under that PEFCR. If such alternatives are unable to provide an equivalent amount of heat, the function of supplying any heat difference between alternatives has to be fulfilled by conventional heat sources, which shall be included in the studied system/scenario and their impacts accounted.

Additional or secondary functions should not be included in the LCA study, and in the related functional unit, if there is no clear evidence that a specific product can realistically provide such functions. This may be the case of electronic devices providing, beyond their primary intended function, negligible amounts of heat compared to the overall demand of buildings where they are used. Any assumptions related to secondary functions attributed to the product shall thus be explicitly documented and subject to the LCA study verification.

Secondary functions shall not be confused with additional functions that a product may provide when it is further used ("reused") downstream in the life cycle after it has fulfilled its primary function, or when its lifespan has been extended. For instance, in some locations, compostable shopping bags may also be used for organic waste collection after their first use for transport of goods, provided they have suitable technical performances for durability and reuse (e.g. by not breaking during product transport) and they can be processed in organic waste treatment facilities (e.g. providing adequate disintegration and biodegradation). Situations of reuse or further use of the product for another function shall be modelled according to the Circular Footprint Formula (CFF; Section 4.4.10.2), under the "Material" part, as also specified in Section 4.4.11<sup>25</sup>.

Similarly to secondary functions, reuse or further use for another function should be considered only where evidence clearly demonstrates it is realistically possible. For instance, this is not the case of compostable shopping bags with unsuitable technical performances to be reused for a second purpose (e.g. organic waste collection), or when the geographical scope of the study includes regions where further use of compostable bags for organic waste collection is restricted (e.g. only dedicated waste bags distributed by municipalities can be used).

If different product alternatives are compared under a specific PEFCR, they shall be all carefully checked for their ability to provide additional functions after reuse or further use, to prevent biased comparisons where such possibility is specifically considered only for a given alternative, potentially disavouring the other ones. For instance, conventional non-biodegradable shopping bags may also be reused for residual waste collection as an additional function, which should be considered for inclusion if reuse of any biodegradable alternative bag is taken into account in the same study.

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<sup>25</sup> This implies identifying a suitable point of substitution where reused compostable shopping bags replaces an equivalent "new" product, and define an appropriate value of the "A" parameter used in the CFF. For instance, reused bags may be assumed to replace new dedicated compostable plastic bags and/or paper bags conventionally provided by municipalities for organic waste collection.

### 3.2.2.2 Functional unit

The functional unit of the LCA study shall describe qualitatively and quantitatively the function(s) of the product(s) and the duration over which they are to be provided, according to the following aspects (further exemplified in Table 3):

- The function(s)/service(s) provided: **"what"**;
- The extent (quantity) of the function or service: **"how much"**;
- The expected level of quality: **"how well"**;
- The duration of the function or service (or the product lifetime if both are the same): **"how long"**;
- The location/geography where the function or service is provided: **"where"**;
- The beneficiary of the function or service (e.g. a consumer/citizen, a professional, a pet animal, etc.): **"for whom"**.

#### Notes:

- The *"how well"* aspect (expected level of quality) may not always be possible to incorporate in the FU. Particularly, subjective and non-quantifiable quality aspects such as fashion, taste shall not be included in the *"how well"* aspect. These relevant quality aspects shall in contrast be reported as aspects that are not captured in the functional unit, but for which equivalence is to be judged by the product user.
- The *"how long"* aspect (duration/life time of the product or number of (re)uses) shall be quantified if technical standards or agreed procedures exist at sectoral level or can be developed. The *"how long"* performance (e.g. number of (re)uses) of the analysed product(s) shall be determined based on evidence, regardless of whether the product is purposefully designed to be reused for a same function (e.g. a multi-use bag) or it is in practice typically reused by final users (e.g. a conventional disposable bag). In this respect, user behaviour and possibly shorter *"how long"* values in reality shall be taken into account in defining the *"how long"* performance of the product; for example, an extremely durable shopping bag may be technically reusable for tens of thousands of times, but such would not reflect use reality. Evidence-based (re)use values shall be applied, instead. For food packaging, the specified duration should not be limited to the shelf-life to be ensured to the packed product, but refer to the total product life before the food/drink is consumed by the final user (and hence extending beyond the shelf life during distribution and retail). The *"where"* aspect is relevant for aspects such as transport distances, applicable environmental legislation and corresponding emission levels (e.g. from electricity production or water supply during product use), energy mix for consumed electricity and heat, technology mix used in processes, typical consumer behaviour during use, available waste collection and treatment systems and pathways (and hence the product's End of Life), and others.
- The *"for whom"* aspect is relevant, for instance, in relation to different consumer subgroups (e.g. adults vs. retirees) which may have different product performance needs or exhibit a different consumer/user behaviour, or to the intended use in private vs. professional settings, which often affects the durability of the product. The *"where"* and *"for whom"* aspects shall not be selected to define an overly narrow or specifically combined scope, which may favour a specific product and be the basis for misleading claims. For instance, if a product is sold in 5 EU countries, the scope shall be the entirety of such countries or one of those, but not a specific combination of only 2 or 3 of them and also not a part of a country only. Similar considerations applies to the *"for whom"* aspect: the entirety of users (and their typical behaviour) or one of the quantitatively relevant user types may be selected, but not a specific combination of user types, nor a non-relevant type or subtype. In any case where the entire scope is not selected (i.e. not all countries and not all user types are included) for both the *"where"* and *"for whom"* aspects, the entire scope shall be identified and

specified. The limited scope shall also be listed in the study limitations section and hence be considered further in the interpretation.

**Table 3.** Example of functional unit definition (LCA of shopping bags, generic).

Aspect	Example
“ <b>What</b> ” (function(s) or service(s) provided)	Carrying of shopping from supermarket to home
“ <b>How much</b> ” (extent of the function or service provided)	An average volume of 22 litres and an average weight of 12 kg of purchased goods
“ <b>How well</b> ” (expected level of quality of the function or service)	Without tearing, puncturing and excessively deforming during the shopping trips
“ <b>How long</b> ” (duration of the function or service / product lifetime)	A minimum of ten times/trips
“ <b>Where</b> ” (location/geography of the function or service)	In the entire EU-28 market
“ <b>For whom</b> ” (beneficiary of the function or service)	By the entirety of consumers

In case applicable standards exist, they shall be used when defining the FU, and cited in the LCA study. If relevant Product Category Rules (PCRs) are available from other schemes, these shall be listed and evaluated. They should be used as a basis when developing a PEFCR, to the extent possible and as long as they don’t contradict the requirements of this method and of the PEF.

In comparative studies (if allowed by the relevant PEFCR), all products shall provide an identical function. The capability of each product to fulfil the specified function should be adequately documented based on technical and/or scientific documentation, or other suitable references (e.g. related to relevant technical properties and performances of the product).

**Note 1:** For intermediate products (e.g. polymers or unspecified plastic parts), the FU is more difficult to define because they can often be converted into a variety of final products, fulfilling multiple functions and whose downstream life cycle is not known. Therefore, a **Declared Unit (DU)** should be applied to such products, and based, for example, on a unit mass (kilogram) or volume (cubic meter) of product<sup>26</sup>. In this case, the reference flow may correspond to the DU.

A fair comparison among intermediate products is only possible if they are identical in all their physical, mechanical, chemical and biological characteristics (and also in their downstream processing, transport, use, End of Life, etc.)<sup>27</sup>. If these conditions are not met, comparison of intermediate products shall not be allowed by any PEFCR possibly developed based on this method.

<sup>26</sup> The concept of *declared unit* is also adopted, for instance, in the International EPD System, for declarations related to (intermediate) products whose further fate and function are not known in terms of use (EPD International AB, 2019).

<sup>27</sup> Technical properties of materials (e.g. mechanical properties) normally determine their ability to provide specific (mechanical) functions or performances, and hence the quantity needed to fulfil the performance requirements specified in the functional unit (i.e. the reference flow). Materials with poorer properties may need to be used in larger quantities to provide a given performance (as far as this performance can be achieved by adjusting the final product dimensions as a function of relevant properties of the specific material). A comparison between alternative intermediate products or materials is hence appropriate only if they are used in the same quantity (mass or volume) in all possible kinds of final application, and if the respective downstream life cycles are also identical. This can only be ensured if the compared materials have the same technical, physical, chemical and biological properties, as reported above.

**Note 2:** For LCA studies of **food packaging products**, the FU shall be defined at the food consumption level (i.e. considering food ready for actual consumption by the final consumer), to allow accounting for packaging (and possibly food) losses occurring also during the *Use* stage (beyond those taking place during *Distribution* and possibly other upstream stages)<sup>28</sup>. See Section 3.2.2.3 on the reference flow for further details.

### 3.2.2.3 Reference flow

An appropriate reference flow shall be determined in relation to the defined functional unit. The quantitative input and output data collected for each lifecycle process and activity included in the analysis shall be calculated in relation to this flow.

**Example of reference flow (LCA of shopping bags, generic):** number of bags (and respective mass) required to carry for ten times the quantity of purchased goods specified in the functional unit, without breaking or damaging during transport by one average consumer in the EU 28.

The LCA study should describe (i) how each aspect of the product performance reflected in the functional unit can affect the environmental performance of the product, (ii) how this effect is included in the LCA calculations and (iii) how an appropriate reference flow is calculated accordingly.

For example, the type of packaging used for a given food product might affect the shelf-life (or even the total life) of the packed food, and hence the amount of food (e.g. salad) actually or potentially wasted at retail and at the *Use* stage. As a consequence, the type of packaging may affect the amount of salad (and thus of packaging) which is needed to fulfil the "*how much*" and "*how long*" aspects of the FU (e.g. "*delivering 1 kg of salad for actual consumption by a final consumer in the EU with an average product life of 7 days*").

The LCA study report should hence describe the potential effects of the specific packaging on food waste, and provide a table with the % of salad and packaging wasted at each lifecycle stage and as a whole (per each type of packaging applied if more alternatives are compared under a specific PEFCR). Finally, the report should describe how the % of salad/packaging waste from the table is integrated in the calculation of the reference flow (i.e. the amount of packaging items and respective packaging material) needed to fulfil the FU. All quantitative input and output data related to the packaging life cycle shall be calculated in relation to this reference flow of packaging needed to deliver 1 kg of consumed salad plus X kg of salad waste.

Note that, if different packaging alternatives are compared under a specific PEFCR, and these involve the generation of substantially different amounts of food loss/waste during *Distribution* and/or *Use* (e.g. due to different product/shelf lives), the LCA results shall be additionally calculated including not only the loss/waste of packaging, but also the contribution of the life cycle<sup>29</sup> of food loss/waste (or at least of the net additional or avoided loss/waste compared to a common reference packaging alternative)<sup>30</sup>. However, estimates of food loss/waste levels associated with specific packaging solutions (in absolute or relative terms) are often unavailable or associated with large uncertainties. Therefore, LCA results including the contribution of the life cycle of food loss/waste shall be presented and discussed separately in the LCA report<sup>31</sup>. Further details on how to

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<sup>28</sup> For instance, the functional unit could be defined as "*delivering 1 kg of a specific food item for actual consumption by the final consumer in Europe*".

<sup>29</sup> Including Raw Material Acquisition and Pre-processing, Manufacturing, Distribution and End of Life, as far as appropriate to the life cycle stage where the loss/waste occurs.

<sup>30</sup> In so far as differences in food loss/waste generation are due to differences in packaging performance (e.g. in the shelf/total life ensured to the product).

<sup>31</sup> Note that the requirement to account for the contribution of product (food) losses/waste in the calculation of the reference flow of packaging items and material required to fulfil the FU (as specified more above in this section) do not prevent to separately calculate LCA results including and excluding the contribution of the life cycle of food losses/waste. As better specified in Sections 4.2.3 and 4.2.4, default LCA calculations and results will only include the life cycle of losses/waste of packaging (as inherently accounted in the

account for packaging and product losses/waste during distribution and at consumer are provided in Sections 4.2.3 (*Distribution* stage) and 4.2.4 (*Use* stage).

Note also that food packaging shall not be seen in itself as a unique possible solution to the food waste problem, while a comprehensive set of solutions is generally needed, targeting causes and consequences of root causes such as overproduction and undervaluing. In this respect, particular care shall be taken in avoiding misleading comparisons and subsequent interpretation of the LCA results when comparing different packaging options for a given food product under a specific PEFCR. This may happen, for instance, when the packaging alternative identified as preferable out of those assessed is claimed as the best delivery option in absolute terms for the product, while an unpackaged variant is available and may be environmentally preferable. If the study intends to claim absolute superiority, the unpackaged variant shall be included in the assessment, otherwise such a claim is not supported by the results and shall not be reported. While this situation is beyond the scope of this specific method, most of the provisions given in this document can be applied to assess also such an additional unpacked alternative. Additional guidance on how to avoid misleading comparisons and interpretation can be found in the ILCD Handbook – Detailed guidance (section 15; EC-JRC, 2010).

### **3.2.2.4 Examples**

Additional examples are provided below on the selection of relevant product functions and subsequent definition of the functional unit and of the corresponding reference flow. Examples refer to selected plastic products from the three most relevant market sectors for plastic applications (i.e. packaging, building & construction and “others”<sup>32</sup>; PlasticsEurope, 2019).

#### **Example 1**

**Product to be assessed:** single-use plastic bottles for carbonated beverages

**Relevant product function:** delivery of generic carbonated beverages to consumers

The **functional unit** can be defined as: “delivering 1 litre of generic carbonated beverage by means of single-use bottles to an average consumer in the EU, without breaking (e.g. collapsing) during transport and ensuring a minimum specified shelf (or product) life for the beverage”

**What:** delivering carbonated beverage

**How much:** 1 litre

**How well:** without breaking (e.g. collapsing) during transport, and ensuring a minimum specified shelf (or product) life for the beverage

**How long:** one time (single-use bottles)

**Where:** in the entire EU-28 market

**For whom:** to an average consumer

**Reference flow:** amount of plastic packaging material needed to fulfil the defined function, which shall be measured in kg of material (including polymer and any additives)

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reference flow). Additional results will then be calculated by also including the life cycle of food losses/waste (or of their net differences compared to a common reference alternative), based on loss/waste rates for the packed product which are consistent with those considered for the calculation of the reference flow (see Section 4.2.3 for further details).

<sup>32</sup> Including appliances, mechanical engineering, furniture, medical devices. Here, furniture is selected as a representative example for this sector.

### **Example 2**

**Product to be assessed:** thermal insulation boards for buildings

**Relevant product function:** provision of thermal insulation to residential buildings

The **functional unit** can be defined as: "providing thermal insulation for 1 m<sup>2</sup> of external wall or roof of residential buildings located in temperate EU regions, ensuring a thermal resistance (R) equal to 0.14 W m<sup>-2</sup>K<sup>-1</sup> throughout a minimum reference service life"

**What:** Providing thermal insulation to buildings

**How much:** for 1 m<sup>2</sup> of external wall or roof

**How well:** with a thermal resistance (R) of 0.14 W m<sup>-2</sup>K<sup>-1</sup>

**How long:** throughout a minimum reference service life (e.g. 50 years, based on the expected average design life of buildings)

**Where:** in temperate EU regions, namely AT, BE, BG, CZ, DE, DK, EE, northern ES, FI, northern and central FR, HU, IE, IS, LI, LT, LU, LV, LX, NL, PO, SE, SI, SK, RO.

**For whom:** for residential buildings

**Reference flow:** amount of insulation material needed to fulfil the defined function, which shall be measured in kg of material (including polymer and any additives)

### **Example 3**

**Product to be assessed:** outdoor stacking chairs

**Relevant product function:** provision of free seating support to people

The **functional unit** can be defined as: "providing free seating support to one person, by means of one chair without armrests and cushioning, with a seat height of 50 cm and which does not break or excessively discolour during outdoor use at households or events over a minimum service life of 10 years"

**What:** Providing a free seating support without armrests and cushioning in outdoor spaces

**How much:** for 1 person with a rated maximum weight of 150 kg (according to standard XY)

**How well:** through stacking chairs with a seat height of 45 cm from the ground, which does not break or excessively discolour during the product lifespan

**How long:** over a minimum service life of 10 years

**Where:** in the entire EU-28 market

**For whom:** for households or private events

**Reference flow:** amount of chair material needed to fulfil the defined function, which shall be measured in kg of material (including polymer and any additives)

### **3.2.3 System boundary**

The system boundary defines which parts of the product life cycle and which associated life cycle stages and processes belong to the analysed system (i.e. are required to carry out its function as defined by the functional unit).

The system boundary shall be defined following a general supply-chain logic for the analysed product(s) and the related functional unit, including all stages from raw material acquisition and pre-processing, through product manufacturing, distribution and storage, the use stage and the end-of-life of the product (i.e. from cradle-to-grave), as appropriate for the intended application of the study. The system boundary shall include all relevant lifecycle processes and activities linked to these stages, except for those processes/activities excluded based on the cut-off rule (see Section 4.6.4). Any deviation from the default cradle-to-grave approach, as well as the reason for and potential significance of any other exclusion shall be explicitly documented and justified.

The co-products, by-products and waste streams of at least the foreground system<sup>33</sup> shall be clearly identified.

For intermediate products (e.g. polymers or unspecified plastic parts), a cradle-to-gate approach is exclusively allowed, i.e. the Use stage and End of Life (see Section 4.2) shall be excluded from the system boundary.

#### **3.2.3.1 System boundary diagram**

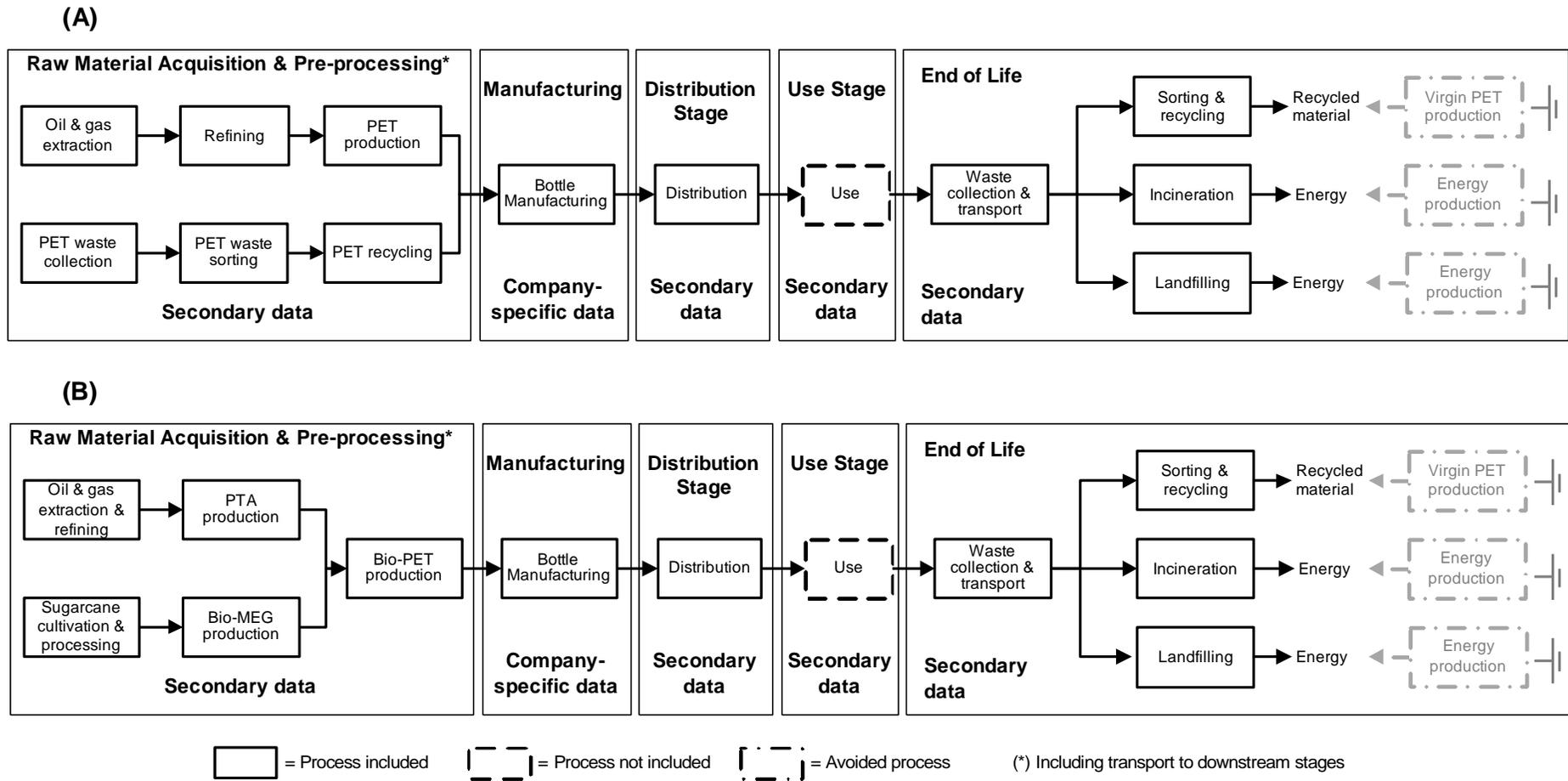
A system boundary diagram (or flow diagram) is a schematic representation of the analysed system. It details which parts of the product life cycle (life cycle stages, processes, and activities) are included or excluded from the analysis. A system boundary diagram can be a useful tool in defining the system boundary and organising subsequent data collection activities.

A system boundary diagram shall be included in the scope definition phase and in the LCA study report. The diagram shall clearly indicate the processes or activities that are included and those that are excluded from the analysis. The names of processes and/or activities in the diagram shall be aligned with those used throughout the LCA report. Processes and activities shall be grouped according to the life cycle stages identified in Section 4.2 (and related nomenclature), i.e. Raw Material Acquisition and Pre-processing, (Product) Manufacturing, Distribution Stage, Use Stage, and End of Life.

The system boundary diagram shall highlight processes and activities where company-specific data are used. Figure 2 provides examples of system boundary diagrams according to the requirements specified above, and considering PET bottles as the product in scope. Requirements on which type of data shall or may be used are provided in the Data Needs Matrix (DNM; Section 4.7.5).

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<sup>33</sup> The foreground system includes the set of foreground processes, i.e. those processes in the product life cycle for which direct access to information is available to the practitioner. These include, for instance, processes run at the producer's site, and other processes operated by the producer or its contractors (e.g. transport of goods, head-office services, etc.).



**Figure 2.** Example of system boundary diagrams conforming to the requirements specified in this method, for the case of partially recycled (A) and partially bio-based PET beverage bottles.

### 3.2.3.2 Indirect effects

In general, indirect effects are here intended as the expected or potential consequences of the investigated product supply chain on other product systems. Such effects may include either market-mediated effects generated by relevant changes in product demand and price, or effects that may not be (easily) put in direct relation with the throughput of a specific supply-chain activity. Even though the quantification of many indirect effects may not be undertaken in a LCA context, or may be subject to a too large level of uncertainty to be used for a reliable assessment, it was considered appropriate to at least mention and briefly evaluate the available information.

One of the most commonly acknowledged indirect effects for bio-based (plastic) products is indirect Land Use Change (iLUC), which is taken into account in this method only for the purpose of providing additional environmental information, i.e. it is not included in the main LCA calculations and results (Section 3.2.5). Further discussion on iLUC effects and recommended quantification methods is reported separately, along with other (direct) land use change effects (see Section 4.4.15.3).

In the case of bio-degradable plastic products, other expected or potential indirect effects reported in the literature include, for instance:

- Effects on the collection and management of other (closely linked) waste streams (e.g. organic waste). Examples in this respect are:
  1. A potential reduction of non-biodegradable plastic impurities in separately collected organic waste when biodegradable waste/shopping bags (or food packaging<sup>34</sup>) are used in place of non-biodegradable ones, and subsequent possible improvement/simplification of downstream biological treatment and better quality of produced compost, (Müller, 2012; Müller and Müller, 2017);
  2. A possible increase in the separate collection rate of household organic waste when biodegradable waste bags are used in place of non-biodegradable ones<sup>35</sup> (Müller, 2012; Müller and Müller, 2017); and
  3. On the opposite, a possible reduction in recycling efficiencies and/or in the quality of recycled conventional plastics due to the presence of contaminating biodegradable and/or non-recyclable plastics (if these cannot be properly sorted out before recycling) (e.g. Soroudi and Jakubowicz, 2013 and Alaerts et al., 2018).
- Effects on crop production systems relying on the use of plastic products. For instance, when proper removal of agricultural mulch film is not performed or cannot be ensured, the shift to a biodegradable material may prevent the accumulation of non-biodegradable plastic in the soil and the subsequent potential decrease of soil quality and of the yield of the affected crop.

Attempts to quantify some of these effects in a LCA context, related challenges, and possible consequences on the LCA results were addressed in a systematic review conducted to inform the development of this method (see Annex K) and are not reiterated here.

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<sup>34</sup> In case non-biodegradable (food-contaminated) packaging was improperly discarded with any food residues along with organic waste.

<sup>35</sup> Although such an increase is most likely a consequence of communication and awareness raising campaigns normally conducted (along with the provision of biodegradable bags) to promote separate collection of organic waste by household, rather than a merit of the material used for organic waste bags. Similar results may also be achieved by means of biodegradable paper bags, provided they are suitable for the area where organic waste is to be collected.

Regarding fossil-based plastics, a number of (potential) indirect effects are discussed in the literature with reference to the supply of fossil-based feedstocks or their displacement with alternative ones (especially focusing on fossil fuels for transport). According to Unnasch et al. (2009) such effects include:

- iLUC caused by agricultural expansion on afforested areas, due to road construction on previously occupied agricultural land for accessing oil fields.
- Emissions and impacts associated with military operations required to protect petroleum supply, as well as impacts from military conflicts to secure access to oil resources (and possible need for reconstruction).
- Effects potentially induced by possible changes in production and subsequent market availability of refinery co-products, due to a reduced fossil fuel demand to refineries when alternative fuels are used. This would lead to decreased crude oil processing and subsequent reduced production and availability of refinery outputs<sup>36</sup>. In the case of replacing fossil-based feedstocks for polymers with alternative ones, such effects may be generated by a reduced demand for naphtha rather than fuels. However, given the flexibility of oil refineries to adjust its product profile, these effects may be entirely or partially compensated by adjusted refinery operation.
- Macro-economic effects due to changes in petroleum usage and price (also referred to as “rebound effect”).

An attempt to quantify the potential impact of such indirect effects has been conducted, for instance, by Malins et al. (2015) limiting to GHG emissions and the resulting climate change impact (as better discussed in Annex F). Based on these estimates, the contribution of each single effect to the total, average climate change impact of fossil-based transportation fuels is in most cases very low (less than 1%), and anyway never exceeding 6% (see Annex F for further details).

Indirect effects such as those reported above (other than iLUC) shall not be included in LCA studies conforming to this method, nor for reporting of additional environmental information. This is because of the even considerable uncertainty, and the high risk of not ensuring a reliable, consistent and reproducible assessment (or non-misleading comparisons under specific PEF CRs).

### **3.2.4 Impact categories and assessment methods**

The environmental performance of the investigated product(s) is evaluated and measured with respect to selected impact categories, representing specific environmental issues of concern. Impact categories selected in this method (Table 4) are therefore comprehensive, in the sense that they aim at covering a broad range of relevant environmental issues for supply chains of plastic products, following the general requirement for completeness for PEF studies (to which this method conforms; see Section 2.4). Covered issues are related to natural resource use and emissions of environmentally damaging substances (e.g. greenhouse gases and toxic chemicals) occurring throughout the life cycle, which may also affect human health. Impact categories are hence directly related to elementary flows between the technosphere and the environment compiled in the Life Cycle Inventory (section 4) and representing direct resource inputs and emissions from the product life cycle.

In the Life Cycle Impact Assessment phase (Section 5), the contribution of each elementary flow to the single impact categories is quantified by means of impact assessment (or characterisation) methods, which apply specific characterisation factors to relevant elementary flows for the category of interest. Characterisation factors are

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<sup>36</sup> For instance, a reduced crude oil processing in refineries would lead to a reduced production and availability of residual oil and petroleum coke, thereby increasing their price. This could in turn lead to a reduced consumption of such refinery outputs, or to a more likely shift to other alternative fuels that can either be “dirtier” (e.g. coal) or “cleaner” (e.g. natural gas), with all the resulting environmental implications.

calculated through specific impact assessment (or characterisation) models, quantifying the environmental mechanism between relevant elementary flows and the potential environmental impact in each impact category. Each category hence relies on a specific impact assessment model and is quantitatively represented by an impact category indicator calculated through specific characterisation factors. Characterised potential impacts are then normalised into a common reference unit (through normalisation factors) and finally weighted and possibly aggregated into a single overall score by means of weighting factors. Life Cycle Impact Assessment results provides the necessary basis for interpretation with respect to the goals of the LCA study (for example, identification of supply chain “hotspots” and options for improvement).

Table 4 provides the list of impact categories and related impact assessment methods (including impact assessment models and related impact category indicators) that shall be applied for the core impact calculations in a LCA study following this method. Listed categories and methods shall be applied without exclusions or additions, and entirely conform to those currently prescribed in the Product Environmental Footprint method. The latter were specifically selected considering the need of ensuring a sufficient robustness of the underlying models, while covering the broadest possible range of impact categories.

The sets of Characterisation Factors (CFs), Normalisation Factors (NFs) and Weighting Factors (WFs) to be used for each impact category are those reported in the most recent version of the EF reference package released at the time of developing the LCA study (currently the EF 3.0 package has been released). The full list of CFs is regularly updated and is available at the following link <http://eplca.jrc.ec.europa.eu/LCDN/developer.xhtml>, which also reports the sets of normalisation and weighting factors to be applied. Users of this method shall specify in the LCA study report the version of the EF reference package used for calculation. Further provisions related to the impact assessment phase are provided in Section 5.

More details on how the CFs were calculated is available at: <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml> (see also Fazio et al., 2018a and Fazio et al. 2018b). For the impact categories *Human Toxicity - cancer*, *Human Toxicity - non-cancer* and *Ecotoxicity - freshwater*, all CFs have been recalculated by the JRC through the USEtox 2.1 model, relying on new input data for physicochemical properties, aquatic ecotoxicity and human toxicity of each substance (see Saouter et al., 2018)<sup>37, 38</sup>. As for the weighting factors, background information is available in Sala et al. (2018).

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<sup>37</sup> CFs calculated according to this technical report shall not be mixed with existing CFs based on USEtox 2.1 (applied until the EF 2.0 reference package) as the method used to calculate some of the input parameters has changed. The report is available at <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>.

<sup>38</sup> Note that some errors have been identified, after the release of the EF 3.0 reference package, in a number of characterisation factors related to the two Human Toxicity impact categories and to freshwater Ecotoxicity. This is currently being addressed within the Data Working Group of the Environmental Footprint (EF) initiative, so that an updated (and possibly reduced) set of factors is expected to be available by the end of the ongoing EF transition phase (2019-2021).

**Table 4.** Default impact categories with respective impact category indicators and impact assessment models that shall be considered and applied in LCA studies conforming to this method. The CFs, NFs and WFs from the latest EF reference package (currently 3.0) that shall be used for each impact category are available at: <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>.

<b>Impact Category</b>	<b>Impact Category indicator</b>	<b>Unit</b>	<b>Impact Assessment Model</b>	<b>Robustness</b>
Climate Change, total <sup>(1)</sup>	Radiative forcing as Global Warming Potential (GWP <sub>100</sub> )	kg CO <sub>2</sub> eq	Baseline model of the IPCC over a 100 year time horizon (IPCC, 2013)	I
Ozone Depletion	Increase of stratospheric ozone breakdown as Ozone Depletion Potential (ODP)	kg CFC-11 eq	Steady-state model of the World Meteorological Organization over an infinite time horizon (WMO, 2014 + integrations)	I
Human Toxicity – cancer <sup>(3)</sup>	Comparative Toxic Unit for humans (CTU <sub>h</sub> )	CTU <sub>h</sub>	USEtox model 2.1 (Fankte et al., 2017)	III
Human Toxicity – non-cancer <sup>(3)</sup>	Comparative Toxic Unit for humans (CTU <sub>h</sub> )	CTU <sub>h</sub>	USEtox model 2.1 (Fankte et al., 2017)	III
Particulate Matter	Impact on human health	Disease incidence	PM method recommended by UNEP (UNEP, 2016)	I
Ionising Radiation – human health	Human exposure efficiency relative to U <sup>235</sup>	kBq U <sup>235</sup> eq	Human Health effect model (Dreicer et al., 1995)	II
Photochemical Ozone Formation - human health	Tropospheric ozone concentration increase	kg NMVOC eq	LOTOS-EUROS model (Van Zelm et al., 2008) as implemented in ReCiPe 2008	II
Acidification	Accumulated Exceedance (AE) of the critical load	mol H <sup>+</sup> eq	Accumulated Exceedance model (Seppälä et al., 2006; Posch et al., 2008)	II
Eutrophication – terrestrial	Accumulated Exceedance (AE) of the critical load	mol N eq	Accumulated Exceedance model (Seppälä et al., 2006; Posch et al., 2008)	II
Eutrophication – freshwater	Fraction of nutrients (P) reaching freshwater end compartment	kg P eq	EUTREND model (Struijs et al., 2009) as implemented in ReCiPe	II
Eutrophication – marine	Fraction of nutrients (N) reaching marine end compartment	kg N eq	EUTREND model (Struijs et al., 2009) as implemented in ReCiPe	II

Impact Category	Impact Category indicator	Unit	Impact Assessment Model	Robustness
Ecotoxicity – freshwater <sup>(3)</sup>	Comparative Toxic Unit for ecosystems (CTUe)	CTUe	USEtox model 2.1 (Fankte et al, 2017)	III
Land Use <sup>(3)</sup>	<ul style="list-style-type: none"> <li>• Soil quality index <sup>(2)</sup></li> <li>• Biotic production</li> <li>• Erosion resistance</li> <li>• Mechanical filtration</li> <li>• Groundwater replenishment</li> </ul>	<ul style="list-style-type: none"> <li>• Dimensionless (pt)</li> <li>• kg biotic production</li> <li>• kg soil</li> <li>• m<sup>3</sup> water</li> <li>• m<sup>3</sup> groundwater</li> </ul>	Soil quality index based on LANCA (Beck et al., 2010 and Bos et al., 2016)	III
Water Use <sup>(3)</sup>	User deprivation potential (deprivation-weighted water consumption)	m <sup>3</sup> world eq	Available Water REMaining (AWARE) as recommended by UNEP, 2016	III
Resource use – minerals and metals <sup>(3)</sup>	Abiotic resource depletion (ADP, based on ultimate reserves)	kg Sb eq	CML 2002 (Guinée et al., 2002) as updated in Van Oers et al. (2002)	III
Resource use – fossils <sup>(3)</sup>	Abiotic resource depletion –fossil fuels (ADP-fossil) <sup>(5)</sup>	MJ	CML 2002 (Guinée et al., 2002) and Van Oers et al. (2002)	III

(1) The indicator “Climate Change, total” consists of three sub-indicators: Climate Change – fossil; Climate Change – biogenic; and Climate Change – land use and land use change. The sub-indicators are further described in Section 4.4.15. The contribution of each sub-indicator shall be reported separately if it is larger than 5% of the total Climate Change score, as also specified in Section 4.4.15.

(2) This index is the result of the aggregation, performed by JRC, of the 4 indicators provided by LANCA model as indicators for land use.

(3) The results of these impact categories shall be interpreted with caution (especially in comparative studies possibly conducted under a specific PEFCR) due to higher uncertainty and lower robustness of the underlying impact assessment methods. This especially applies to *Human Toxicity–cancer*, *Human Toxicity–non-cancer*, and *Ecotoxicity–freshwater* (where uncertainty can be up to 1-2 orders of magnitude) and to a lower extent also to *Water Use*, and *Land Use*. Moreover, results calculated for *Resource Use–minerals and metals* after normalization may be overestimated. In this respect, the European Commission intends to develop a new method moving from depletion to dissipation model to better quantify the potential for conservation of resources.

(4) Addressing potential impacts from direct occupation and transformation of aboveground land from any kind of activity, including underground mining (e.g. for oil exploitation).

(5) In the EF elementary flow list, and for the current recommendation, Uranium is included in the list of energy carriers, and it is measured in MJ.

### 3.2.5 Additional information to be included in the LCA study

Relevant potential environmental impacts of a product may go beyond the default life-cycle-based impact categories considered in this method. It is important to consider and separately report such additional impacts as complementary “additional environmental information”, whenever feasible, relevant and non-misleading.

For example, biodiversity impacts may occur in a specific site or multiple locations across the life cycle, due to human-induced land use changes and subsequent occupation, crude

oil drilling, extraction and transport<sup>39</sup>, or other supply-chain activities such as transport and related infrastructure. Addressing these impacts may require the application of further differentiated life cycle impact assessment (LCIA) methods for the *Land Use* category, or of additional impact categories (with the underlying LCIA methods) that are not included in the default list provided in this guide (Section 3.2.4). Additional qualitative descriptions may also need to be applied where impacts cannot be linked to the product supply chain in a quantitative manner.

Potential biophysical/biodiversity impacts on (aquatic) ecosystems or aesthetic/landscape impacts from any kind of plastic product released to the open environment (due to e.g. littering) are also currently not captured within existing impact categories<sup>40</sup>. The same holds true for toxicological impacts on marine and terrestrial ecosystems of (chemical) compounds from fragmentation and (bio)-degradation of plastic products in the open environment, including micro- and nano-plastics (default impact categories do not include marine and terrestrial ecotoxicity)<sup>41</sup>. Additional categories addressing quantitatively, qualitatively or semi-quantitatively the biophysical or landscape impacts of littered plastic products (e.g. physical damage to wildlife or aesthetic effects) may hence be included as additional information. Similarly, one or more additional categories may be included to address toxicological/chemical impacts from fragmentation and (bio)-degradation of plastic products, as long as these do not overlap with default toxicity-related impact categories covered in this method (and as far as better knowledge is gained on the fate, exposure and effects of plastics released into the environment. It is noted that there is an ongoing project (MarILCA)<sup>42</sup> aiming at integrating marine litter impacts in LCA, which may provide useful impact assessment methods and indicators to quantitatively address at least a part of relevant potential impacts from plastic products littered in the marine environment.

Some products might be produced in companies that are located close to the sea, and directly emitting into this environmental compartment. More in general, specific product supply chains may include activities involving direct emissions to seawater (e.g. of toxic substances), which might directly impact marine ecosystems. For example, offshore oil extraction activities may contribute to such impact with direct emissions from drilling mud (to seabed), from oxidation of sacrificial anodes on pipelines, and from other auxiliaries used in large quantities offshore (e.g. paint debris from regular maintenance required in corrosive conditions)<sup>43</sup>. While the default set of impact categories considered in this method only address ecotoxicity impacts on fresh water ecosystems, it is important to also consider emissions that occur directly into marine water and their impact on marine ecosystems. For the time being, these emissions need to be included at the elementary flow (i.e. at the life cycle inventory) level, because no sufficiently reliable impact assessment model is currently available to quantify potential toxicity impacts on marine ecosystems.

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<sup>39</sup> Due for instance to oil losses occurring during these operations, as far as they can be considered "structural" losses associated with normal/average production and supply conditions, while accidental losses (due to accidents or disasters) are typically excluded from LCA and PEF studies.

<sup>40</sup> Note that this applies not only to non-biodegradable plastic products, but also to biodegradable products over the (more or less long) timeframe between their release and (full) biodegradation in the open environment.

<sup>41</sup> Impacts on freshwater ecosystems due to the release of several chemicals are instead covered in the "Ecotoxicity - freshwater" impact category, but micro- and nano-plastics are not covered, as well as several plastic additives that may be released during degradation. Moreover, a complete identification and quantification of emissions from plastics fragmentation and (bio)-degradation in the open environment is currently difficult, due to incomplete knowledge about fragmentation and degradation pathways. This leads to incomplete Life Cycle Inventories of the resulting emissions into the environment.

<sup>42</sup> The MarILCA project (<https://www.mariica.org/>) is a joint effort of the UN Life Cycle Initiative and the Forum for Sustainability through Life Cycle Innovation (FSLCI).

<sup>43</sup> Seawater emissions from these sources are frequently omitted from life cycle inventory datasets related to crude oil supply, but their relevance should be checked and their contribution included as appropriate when developing new datasets or updating existing ones (see also Section 4.4.1).

Beyond additional environmental issues and impacts mentioned above, relevant technical aspects and/or physical properties of the product in scope may need to be considered. These aspects shall be reported as additional technical information.

Section 3.2.5.1 specifies which additional environmental information shall, should or may be included, while Section 3.2.5.2 provides requirements for additional technical information.

### **3.2.5.1 Additional environmental information**

Additional environmental information shall be:

- Based on information that is substantiated and has been reviewed or verified in accordance with the requirements of ISO 14020:2000 and Clause 5 of ISO 14021:2016;
- Specific, accurate and not misleading;
- Relevant to the particular product category;
- Life cycle based information additional to the default impact categories (listed in Section 3.2.4).

Additional environmental information shall only be related to environmental aspects. Information and instructions (e.g. product safety sheets), which are not directly related to the environmental performance of the product shall not be part of additional environmental information.

Additional environmental information shall not reflect the same or similar impact categories as those considered by default in this method (Section 3.2.4), shall not substitute the underlying characterisation models, and shall not report any results obtained by applying any new CFs to such impact categories. Moreover, additional environmental information shall not include information that is already fully or mostly considered in the life cycle model (e.g. any recycled content in the product, incorporation of bio-based material, specific product/packaging characteristics such as compostability, recyclability and applicability of dedicated take-back schemes, etc.).

Additional environmental information shall be presented separately in the LCA study report and should include (closed list):

- (a) Potential impacts due to indirect Land Use Change (iLUC) on Climate Change and possibly other relevant impact categories (e.g. Eutrophication, Acidification, Land Use and Water Use), from any relevant type of land demand occurring throughout the product life cycle. GHG emissions from iLUC shall be calculated according to the provisions reported in Section 4.4.15.3, and the corresponding contribution to the Climate Change indicator shall be reported separately as a net result. Any specific iLUC mitigating measures implemented by the company undertaking the LCA study may also be reported.
- (b) Quantitative, qualitative or semi-quantitative information related to potential impacts of the product in scope on biodiversity (and any specific mitigating measures implemented by the company undertaking the LCA study). Note that there shall be no overlap with potential impacts covered by the default impact categories adopted in this method (e.g. Climate Change, Land Use, Water Use, Ecotoxicity etc.; see Table 4). A list of biodiversity indicators that may be applied is provided in Section 3.2.5.1.1.
- (c) Estimated contribution of the product in scope to plastic litter (macro-plastic) formation and release in the open environment (terrestrial, riverine and/or marine) at End of Life (expressed per functional unit). Quantification may be conducted according to the method described in Section 4.4.10.12 and Annex I (i.e. the *PLP method*; Peano et al., 2020).

- (d) Estimated contribution of the product in scope to secondary micro- and nano-plastics generation and release in the open environment (terrestrial, riverine and/or marine) at End of Life and throughout the whole upstream life cycle (expressed per functional unit). Quantification may be conducted according to the method described in Section 4.4.10.12 and Annex I (i.e. the *PLP method*; Peano et al., 2020).
- (e) Quantitative, qualitative or semi-quantitative information on potential environmental impacts from littering and/or release of the product in scope as macro-plastic in the open environment (e.g. bio-physical, toxicological or aesthetic/landscape impacts). While no impact assessment methods are currently available to be prescribed or recommended, they may become available in the future and may be applied for quantification.
- (f) Quantitative, qualitative or semi-quantitative information on potential impacts from secondary micro- or nano-plastics released or generated throughout the whole product life cycle<sup>44</sup>. While no impact assessment methods are currently available to be prescribed or recommended, they may become available in the future and may be applied for quantification.

Additional environmental information may include (closed list):

- (a) Information on local/site-specific impacts;
- (b) Offsets (limited to GHG emissions, according to the provisions reported in Section 4.4.16);
- (c) Noise impacts;
- (d) Amount (per functional unit) of biogenic and non-biogenic carbon in the product that is not released (e.g. not mineralised) after 100 years<sup>45</sup> from disposal (i.e. landfilling) or other final End of Life treatments/fate (e.g. on-land application of residual organic material from composting/digestion, or in-situ biodegradation);
- (e) Soil carbon uptake (accumulation) from e.g. improved land management through tilling techniques or from grasslands (if proof is provided), expressed per functional unit (see Section 4.4.15.4).

Overall emissions of substances released directly into marine water shall also be included in the additional environmental information. Reporting shall be made at the elementary flow level and separately for each substance, as an annex referenced from the section of the LCA report with results of additional environmental information.

The supporting models of any additional information shall be clearly referenced and documented together with the corresponding indicators.

If additional environmental information is used to support the interpretation phase of a LCA study, then all data needed to produce such information shall be assessed against the same quality requirements established for the life cycle inventory data used to calculate the LCA results for the default impact categories covered in this method (see Section 4.7).

### **3.2.5.1.1 Biodiversity**

In line with the PEF method, this method does not include any impact category explicitly covering potential biodiversity impacts (i.e. named as "Biodiversity"), as currently there is no international consensus on a life cycle impact assessment method capturing such impacts within a single impact category. Similarly, there is no consensus on how to

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<sup>44</sup> Ensuring there is no overlapping with any previously estimated impact from micro- and nano-plastics generated via product littering/release as macro-plastic in the open environment, which add to micro- and nano-plastic generated from other specific sources throughout the supply chain.

<sup>45</sup> The 100 years count from the time the product enters the End of Life stage. For instance, for long-living construction products, this may be 150 years from the time of resource extraction and manufacturing.

possibly capture the combined effect of established impact assessment methods used for individual impact categories contributing to a potential impact on biodiversity. However, this method includes at least eight impact categories that have a key effect on biodiversity (i.e., Climate Change, Eutrophication – freshwater, Eutrophication – marine, Eutrophication – terrestrial, Acidification, Water Use, Land Use, Ecotoxicity – freshwater). Additionally, Photochemical Ozone Formation and (stratospheric) Ozone Depletion directly and relevantly impact plant and animal life.

Considering the variability of how biodiversity is impacted by different product groups, biodiversity should also be addressed separately, in addition to the default impact categories, despite these broadly cover a number of “midpoint” impact drivers relevant to biodiversity. Each LCA study shall explain whether such otherwise not covered biodiversity aspects are relevant for the product in scope. If that is the case, the user of this method shall include biodiversity indicators under additional environmental information.

The following options may be applied (closed list) to cover biodiversity as additional environmental information:

- Report the percentage of material that comes from ecosystems that have been managed to maintain or enhance conditions for biodiversity, as demonstrated by regular monitoring and reporting of biodiversity levels and gains or losses (e.g. less than 15% loss of species richness due to disturbance, but the studies may set their own level provided this is well justified and not in contradiction to a relevant existing PEFCR, if any). The assessment should refer to materials that end up in the final product and to materials that have been used during the respective production process. Examples of such materials are sugarcane, maize or other crops used as feedstock to produce plastic precursors, charcoal that is used in steel production, or soy that is used to feed cows that produce dairy, etc.
- To report additionally the percentage of such materials for which no chain of custody or traceability information can be found.
- To use a certification system as a proxy. The user of this method should determine which certification schemes provide sufficient evidence for ensuring biodiversity maintenance and describe the criteria used. A useful overview of standards is available on <http://www.standardsmap.org/>.

### **3.2.5.2 Additional technical information**

Additional technical information may include (closed list):

- (a) Bill of materials data;
- (b) Information of dismantleability or reparability;
- (c) Information and data related to the technical performance of the product relevant to the functional unit.

When the product in scope is an intermediate product, additional technical information shall include:

- (a) The biogenic and non-biogenic (e.g. fossil) carbon content at factory gate (physical content and allocated content);
- (b) Recycled content ( $R_1$ ) of the product<sup>46</sup>, (followed by an explicit clarification statement that the environmental implications of using recycled material are already fully considered in the LCA results);

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<sup>46</sup> Recycled material from reprocessing of post-consumer plastic waste should be considered to define the recycled content ( $R_1$ ) of a product, while material derived from pre-consumer scrap circulating within a process chain or pool of process chains should be generally excluded. Exceptions may apply in specific situations, to be evaluated on a case-by-case basis and to be submitted to the LCA study verification.

- (c) Results with application-specific A-values of the Circular Footprint Formula, if relevant.

### **3.2.6 Assumptions/limitations**

In LCA studies, several limitations to carrying out the analysis may arise and therefore assumptions need to be made. For example, applied secondary data may not completely represent the reality of the analysed product supply chain (e.g. in terms of technology or geography) and it may have not been possible to adapt them for better representation.

All limitations and assumptions shall be transparently reported in the LCA study, and be explicitly taken into account in the interpretation as to their relevance for the results and conclusions, if any.

Limitations related to the following aspects shall be specifically reported:

- Degree of analytical depth and rigour of the study (Section 3.1);
- Selection of a restricted scope as for the “where” and “for whom” aspects of the FU (i.e. not all countries of use and not all user types are considered for the product in scope but only one of them) (Section 3.2.2.2);
- Use of proxy (EF compliant) datasets to represent specific processes in the product life cycle (Section 4.6.3);
- Exclusion of specific processes from the LCI model (due to absence of suitable EF compliant or ILCD-Entry Level compliant proxy datasets), i.e. data gaps.

Other limitations may affect the modelling of the product life cycle, and should be additionally reported in the LCA study. For example, limitations related to the following aspects should be reported:

- Assumptions on the real configuration of the investigated supply chain(s);
- Data and parameters used to define the functional unit and calculate the corresponding reference flow;
- Data and parameters used to calculate process and/or material quantities;
- Assumptions, data and parameters used to model specific processes or activities in the supply chain (e.g. process yields, quantified energy/material inputs and emissions, applied technology(ies) and/or practices, geographical scope etc.);
- Assumptions and data applied to define end of life scenarios (applied pathways and related shares);
- Assumptions, data and parameters used to model end of life processes (e.g. product composition, degradation rates, process efficiencies, technological parameters, other product-specific or waste-specific parameters, etc.).

## 4 Life Cycle Inventory

An inventory of all material and energy inputs, waste outputs, and emissions into air, water and soil throughout the whole product life cycle shall be compiled as a basis for calculating the potential environmental impacts of the analysed product(s). This is referred to as Life Cycle Inventory.

Detailed data requirements and quality requirements to compile the Life Cycle Inventory are described in Sections 4.6 and 4.7.

The Life Cycle Inventory shall adopt the following classification of the flows included:

- **Elementary flows**, which are “material or energy entering the system being studied that has been drawn from the environment without previous human transformation, or material or energy leaving the system being studied that is released into the environment without subsequent human transformation.” (ISO 14040:2006, 3.12). Elementary flows are, for example, resources extracted from nature or emissions into air, water, soil. They are directly linked to the characterisation factors of default (EF) impact categories via impact assessment models (as better described in Section 5);
- **Non-elementary (or complex) flows**, which are all the remaining inputs (e.g. electricity, materials, transports) and outputs (e.g. waste, by-products) in a system, which require further modelling efforts to be transformed into elementary flows.

All non-elementary flows in the Life Cycle Inventory shall be transformed into elementary flows, apart from the product flow of the product in scope. For example, waste flows shall not only be reported in the study as kg of household or hazardous waste, but shall be inventoried considering the emissions into water, air and soil and resource consumption due to the treatment or disposal of such waste (via e.g. incineration or landfilling)<sup>47</sup>. The compilation of the Life Cycle Inventory is therefore only completed when all non-elementary flows are expressed as elementary flows, with exclusively the final product or service flow remaining in the inventory (i.e. the “reference flow” to which all other data quantitatively relate). If developed, the LCI dataset including the overall product inventory (i.e. the inventory of the whole LCA study) shall only contain elementary flows, apart from the product flow of the product in scope.

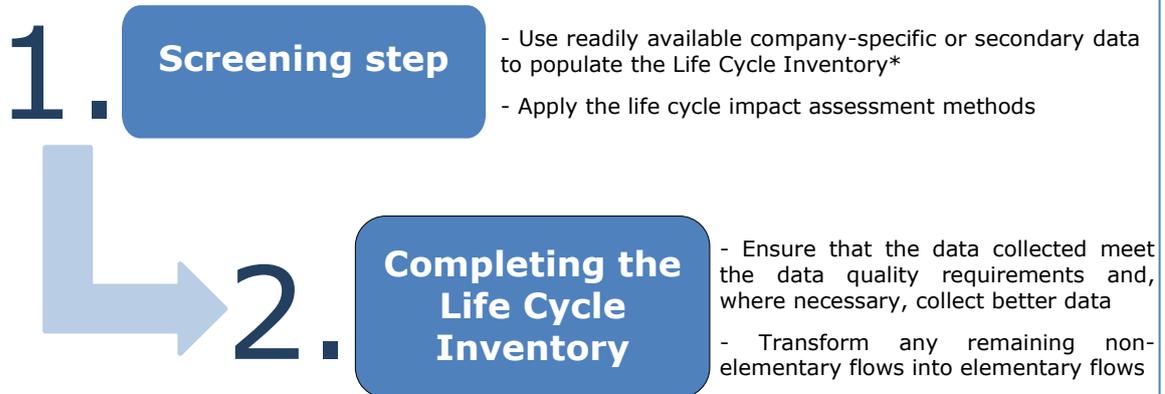
Compiling the Life Cycle Inventory of a LCA study should be completed following a 2-step procedure, as detailed in Figure 3. The first step (“screening step”) is not mandatory, but is highly recommended, because it helps to focus data collection activities and data quality priorities for the final Life Cycle Inventory. Further requirements on how to conduct the screening step are provided in Section 4.1.

**TIP:** Documenting the data collection process is useful to improve data quality over time, facilitate the verification, and to revise future product inventories to reflect changes in production practices.

<sup>47</sup> Exclusively a limited set of radioactive waste flows may be included as such in the inventory, as there is no widely agreed and practice-tested models or data available to model radioactive waste deposition. The waste flows of the latest EF flow list shall be used for this purpose.

# Life Cycle Inventory

## Recommended two-step procedure



(\*) Fulfilling data quality requirements to the extent possible and applying no cut-off

**Figure 3.** Two-step procedure that should be applied to compile the Life Cycle Inventory.

### 4.1 Screening step

An initial “screening” step is highly recommended because it helps to focus data collection activities and data quality priorities for the final Life Cycle Inventory. A screening step shall include the LCIA phase and allow to further refine the life cycle model of the product(s) in scope in an iterative way, as more information becomes available.

All relevant processes and activities in the product life cycle shall be included in the screening step, and no cut-off is allowed. Readily available company-specific or secondary data may be used for the screening step, fulfilling the data quality requirements defined in Section 4.7 to the extent possible.

After the screening step is performed, the initial scope settings may be refined. Any exclusion of life cycle stages or processes based on the screening step shall be explicitly documented and justified, and the respective influence on the LCA results shall be discussed. Any exclusion shall be submitted to the verification process, where conducted.

### 4.2 Life cycle stages

The following default life cycle stages shall be considered, as a minimum, for inclusion in the Life Cycle Inventory of final products:

- Raw Material Acquisition and Pre-processing (including agricultural and forestry production, as well as production of parts and unspecified components);
- Manufacturing (production of the main product);
- Distribution (covering product distribution, storage and related logistics);
- Use stage;
- End of Life (including product recycling or recovery).

In case the naming of the default life cycle stages is changed, the practitioner shall specify which default life cycle stage it corresponds to.

If justified, the user of this method may decide to split or add life cycle stages. The justification shall be included in the LCA report. For example, the life cycle stage 'Raw

Material Acquisition and Pre-processing' may be split into 'Raw Material Acquisition', 'Pre-processing', and 'Transport of processed materials or parts'.

For intermediate products (e.g. polymers or unspecified plastic parts), the following life cycle stages shall be excluded:

- Use stage;
- End of life (including product recycling or recovery).

#### **4.2.1 Raw material acquisition and pre-processing (Cradle-to-gate)**

The "*Raw Material Acquisition and Pre-processing*" stage starts when resources are extracted from nature and ends when the product components enter (through the gate of) the product's manufacturing facility. Processes that may occur in this stage include (non-exhaustive list):

- Mining and extraction of resources (e.g. oil and natural gas);
- Pre-processing of all material inputs to the studied product, such as:
  - Oil refining and downstream processing or relevant refinery outputs;
  - Ore concentration, reduction and forming of metals into ingots;
  - Cleaning coal;
  - Sorting/pre-processing and recycling of waste materials providing recycled content to the product;
- Agricultural and forestry activities (including plant photosynthesis, where relevant) and subsequent conversion/processing to feed or food ingredients, basic wood parts, or plastic precursors/monomers;
- Transportation within and between extraction and pre-processing facilities, and to the production facility.

More specific provisions and examples of processes and activities to be included under the "*Raw Material Acquisition and Pre-processing*" stage in the case of plastic products are reported in Sections 4.2.1.1-4.2.1.5, considering the different types of feedstock that may be used for plastic production. These activities may be split into two separate and more disaggregated stages, such as "*Feedstock Supply*" and "*Polymer Production*".

Production of packaging used for the product in scope shall be modelled as part of the "*Raw Material Acquisition and Pre-processing*" life cycle stage. However, when packaging is the product in scope, the respective production shall be part of the Manufacturing stage (Section 4.2.2).

Treatment or disposal of any waste flow generated during raw material acquisition and pre-processing shall be included in the modelling of the "*Raw Material Acquisition and Pre-processing*" stage. The Circular Footprint Formula (CFF) shall be applied to such waste (Section 4.4.10.2).

##### **4.2.1.1 Fossil-based plastic products**

For **fossil-based plastic products**, the following processes and activities shall be included under the "*Raw Material Acquisition and Pre-processing*" stage, as far as relevant to the specific supply chain investigated:

- Oil and natural gas exploration, drilling and extraction;
- Oil refining into naphtha or any other relevant hydrocarbon used downstream in the supply chain;
- Cracking of Naphtha into relevant monomer(s) (e.g. Ethylene, Propylene or Butadiene) or other relevant intermediates/precursors used downstream in the supply chain;

- (Catalytic) reforming of Naphtha or natural gas;
- Conversion of any intermediates/precursors into the relevant monomer(s) or final precursor(s) used for polymer production;
- Polymerisation and possible compounding (including production and emissions of any used additives, whether for colouring, performance improvement, or process aiding);
- Transport within and between all the listed activities.

#### **4.2.1.2 Plastic waste-based (recycled) products<sup>48</sup>**

For plastic products based on post-consumer plastic waste, processes and activities to be included under the “*Raw Material Acquisition and Pre-processing*” stage are defined by the Circular Footprint Formula (CFF; Section 4.4.10.2), which shall be applied to model recycling situations (Section 4.4.2). The following processes and activities shall thus be included, as far as relevant to the specific supply chain investigated (and in a quantity/share depending on the value of the CFF parameters):

- Collection and transport of post-consumer plastic waste from relevant sources;
- Sorting, screening and/or pre-treatment of collected post-consumer plastic waste;
- Recycling/Reprocessing of sorted, screened and/or pre-treated plastic waste into new polymer granulate, polymer solution, monomers, liquid/gaseous feedstock, or intermediate plastic products (depending on where substitution of virgin with recycled materials or products actually takes place)<sup>49</sup>, including any purification, isolation and separation step, as well as production and emissions of any used additives (whether for colouring, performance improvement or process aiding);
- Transport within and between all the listed activities.

Additionally, the processes and activities related to raw material acquisition and pre-processing of the actually replaced virgin material or product shall be included, until the point of substitution and in a quantity/share depending on the CFF parameters. For example, in case of recycled polymer granulate replacing the corresponding virgin granulate from fossil feedstock, the processes and activities reported in Section 4.2.1.1 shall be included in the “*Raw Material Acquisition and Pre-processing*” stage.

Note that analogous provisions to those reported above apply to waste (end of life) plastic products or parts that are refurbished for reuse or further use in a product with different specifications/function. Conversely, they do not apply in case of direct looping of refillable bottles, crates, or other packaging items, which is modelled differently, outside the CFF.

#### **4.2.1.3 Bio-based plastic products from primary biomass sources**

For bio-based plastic products, the following processes and activities shall be included under the “*Raw Material Acquisition and Pre-processing*” stage, as far as relevant to the specific supply chain investigated:

- Land clearing and related land transformation (land use change) burdens<sup>50</sup>;
- Crop/Plant (biomass) cultivation<sup>51</sup>;

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<sup>48</sup> Note that the processes and activities reported in this section are identical to those that shall be included in the End of Life stage when a product is recycled, while a share of such End of Life processes/activities is included in the modelling of recycled content, reused parts, etc., under the *Raw Material Acquisition and Pre-processing* stage.

<sup>49</sup> For instance, in case of mechanical recycling into new polymer granulate, the following activities shall be included, as far as relevant to the specific recycling process applied: grinding/shredding, water-based density separation, washing, and extrusion.

<sup>50</sup> For GHG emissions due to land use change, the provisions reported in Section 4.4.15.3 shall be applied.

<sup>51</sup> Including production of seeds and seedlings, unless its exclusion is allowed based on the cut-off criteria (see Section 4.6.4).

- Biomass processing into the relevant intermediate(s)/precursor(s) (e.g. sugarcane milling and fermentation to bioethanol and wet milling of maize into starch);
- Conversion of intermediate(s)/precursor(s) into the relevant monomer (e.g. bioethanol to bio-Ethylene, starch into glucose and lactic acid);
- Polymerisation and possible compounding (including production and emissions of any used additives, whether for colouring, performance improvement or process aiding);
- Transport within and between all the listed activities.

#### **4.2.1.4 Products derived from bio-based waste and by-products<sup>52</sup>**

For plastic products derived from bio-based waste and by-products, the components of the “*Raw Material Acquisition and Pre-processing*” stage are defined by the Circular Footprint Formula (CFF; Section 4.4.10.2), which shall be applied to model the use of such feedstock materials for polymer production (see Section 4.4.4). Processes and activities that shall hence be included in this stage, depending on the specific feedstock used, are:

- Possible collection, handling (e.g. baling) and transport of the waste or by-product from relevant sources<sup>53</sup>;
- All (subsequent) processing, conversion, handling, storage, and transport activities carried out after the waste or by-product is generated or collected<sup>54</sup>, until it is used to replace an alternative equivalent substance or material<sup>55</sup> regularly produced from primary resources (i.e. until the point of substitution);
- Any further conversion process into intermediates and/or monomers;
- Polymerisation and possible compounding (including production and emissions of any used additives, whether for colouring, performance improvement or process aiding);
- Transport within and between all the listed activities.

The processes and activities related to raw material acquisition and pre-processing of the replaced alternative material shall also be included, until the point of substitution and in a quantity/share depending on the CFF parameters.

#### **4.2.1.5 CO<sub>2</sub>-based plastic products**

Within the current industrial and market conditions, raw gaseous CO<sub>2</sub> released from point emission sources shall be considered as a waste for recycling (with recycling taking place through capture and conversion into a useful CO<sub>2</sub>-based product; see Section 4.4.5). The use of CO<sub>2</sub> as a feedstock for polymers production shall hence be modelled via the Circular Footprint Formula (CFF; Section 4.4.10.2), which shall be applied in any recycling situation and defines the upstream system boundary. The following process and activities shall thus be included in the “*Raw Material Acquisition and Pre-processing*” stage, as far as relevant to the specific supply chain:

- CO<sub>2</sub> capture/extraction, purification, compression/liquefaction and transport to downstream utilisation processes<sup>56</sup>;

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<sup>52</sup> Including products stemming from joint production/generation with other (main) products, which have a zero or negative economic/market value at the point where they occur (point of arising). For bio-based products that have a positive economic value at the point of arising, the provisions given in Section 4.2.1.3 apply.

<sup>53</sup> These processes and activities will be included in a quantity/share depending on the value of the CFF parameters.

<sup>54</sup> These activities will be included in a quantity/share depending on the value of the CFF parameters.

<sup>55</sup> Intermediate, monomer or polymer.

<sup>56</sup> These processes and activities will be included in a quantity/share depending on the value of the CFF parameters.

- Conversion of captured CO<sub>2</sub> into the first useful CO<sub>2</sub>-based product replacing an equivalent product from primary resources as an input to a same identical process (e.g. CO<sub>2</sub>-based propylene or polyols)<sup>57</sup>;
- Any subsequent conversion process into further intermediates and/or monomers;
- Polymerisation and possible compounding (including production and emissions of any used additives, whether for colouring, performance improvement or process aiding);
- Transport within and between all the listed activities.

The processes and activities related to raw material acquisition and pre-processing of the replaced equivalent product shall also be included, until the point of substitution and in a quantity/share depending on the CFF parameters. For example, in the case of CO<sub>2</sub>-based propylene or polyols replacing their fossil-based equivalents, all supply chain processes and activities involved in the production of the latter shall be included in the “*Raw Material Acquisition and Pre-processing*” stage (as reported in Section 4.2.1.1).

#### **4.2.2 Manufacturing**

The product *manufacturing* stage begins when the product components or constituting material(s) enter the manufacturing site(s) and ends when the finished product leaves the (final) manufacturing facility. Examples of manufacturing-related activities include (non-exhaustive list):

- Chemical processing;
- Manufacturing;
- Transport of semi-finished products between manufacturing processes;
- Assembly of parts, components, or subassemblies.

For plastic products, the *Manufacturing* stage typically includes the conversion of raw polymer granulate (from polymerisation or compounding) into a finished item with a defined shape and size. For complex polymer-based products, these steps may however be part of the preceding “*Raw Material Acquisition and Pre-processing*” stage. Common conversion processes applied to plastic polymers are injection moulding, different versions of blow moulding (e.g. stretch-blow moulding or extrusion-blow moulding), extrusion (e.g. blown film extrusion) and thermoforming. Any assembly activity of products made of different plastic parts shall also be included in the *Manufacturing* stage, as well as production and emissions of any additives used during polymer conversion (whether for colouring, performance improvement or process aiding). Note that more than one company or site may be involved in transforming pre-processed raw materials (i.e. intermediate products) into final products. Transport and transfer activities among different sites/companies (and where relevant also within single sites/companies) shall be included.

Treatment or disposal of any waste flow generated during product manufacturing shall be included in the modelling of the *Manufacturing* stage. The Circular Footprint Formula shall be applied to such waste (Section 4.4.10.2).

#### **4.2.3 Distribution stage**

Products are distributed to users and may be stored at various points along the supply chain. The *Distribution* stage includes transport from the factory gate (manufacturing site) to wholesale warehouses and/or retail stores, storage at these premises, and transport from the latter to the place of use or consumption (e.g. consumer home). Examples of processes and activities related to distribution and storage that shall be included in the *Distribution* stage are (non-exhaustive list):

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<sup>57</sup> Conversion processes will be included in a quantity/share depending on the value of the CFF parameters.

- Energy inputs for warehouse lighting, heating, and refrigeration (where relevant; see the paragraph immediately below);
- Use of refrigerants in warehouses and vehicles (where relevant; see below);
- Fuel use by vehicles;
- Roads construction and maintenance and trucks life cycle (production of materials, manufacturing and End of Life), unless excluded under the cut-off rules (Section 4.6.4).

For LCA studies of plastic products used as packaging for items requiring refrigerated storage (e.g. food items), the burdens from refrigeration at warehouses or retail stores shall not be included in the inventory (being totally assigned to the packaged product). Only burdens related to other relevant activities carried out at such premises, such as lighting and heating, shall be accounted. However, if alternative packaging solutions are compared under a specific PEFCR, and these imply significantly different refrigeration requirements for the packaged product (e.g. due to significantly different product shelf-lives), refrigeration shall be included (at least in terms of net difference compared to a reference alternative).

Collection, treatment and/or disposal of any waste flow generated during distribution and storage shall be included in the *Distribution* stage. The Circular Footprint Formula shall be applied to such waste. Loss of the product(s) in scope occurring during distribution and storage shall also be included in the modelling of the *Distribution* stage, accounting for the burdens from end of life collection/treatment/disposal of such product loss, and those occurring in the *Distribution* stage until the point of loss generation. Upstream burdens associated with raw material acquisition and manufacturing of product loss shall instead be accounted in the relevant life cycle stages (i.e. *Raw Material Acquisition and Pre-processing* and *Manufacturing* stages). Default loss rates per type of product during distribution (and at consumer) are provided in Annex B and shall be used in case no specific and more representative information is available. Allocation rules on energy consumption at storage are presented in Section 4.4.8, while for transport see Section 4.4.7.

For LCA studies of packaging products (e.g. food packaging), losses and/or waste during distribution and storage shall be normally accounted only for the packaging, and not also for the respective product content (e.g. food). However, if alternative packaging solutions with substantially different loss or waste rates are compared under a specific PEFCR, LCA results shall be additionally calculated also including the production<sup>58</sup> and End of Life (i.e. the life cycle) of losses/waste of the packaged product (or at least of the net additional or avoided loss/waste compared to a common reference alternative)<sup>59</sup>. In this case, the life cycle of losses/waste of packaged product occurring during use (or of their net difference from a common reference) shall also be considered, in the *Use* stage, as specified in section 4.2.4. Estimates of (relative) food loss/waste rates associated with specific packaging solutions are often unavailable, and normally affected by large uncertainties. Moreover, for food losses/waste occurring at the consumer level, the contribution of the used packaging acts in parallel with that of consumer's behaviour, and may not be easily distinguished. Therefore, the LCA results including the contribution of the life cycle of losses/waste of packaged product (e.g. food) shall be presented and discussed separately in the LCA report. For such additional calculations, default loss rates reported in Annex B cannot be normally applied, as they are packaging-unspecific and often refer to broad product categories (e.g. for food), so that they are unsuitable to capture possible differences among alternative packaging solutions for specific (food) products. Alternative rates should hence be estimated or assumed, and these shall be consistently applied both in calculations including the contribution of losses/waste of the packaged

<sup>58</sup> Including Raw Material Acquisition and Pre-processing, Manufacturing and Distribution until the point where the loss/waste occurs.

<sup>59</sup> In so far as differences in loss/waste rates are due to differences in packaging performance (e.g. in the shelf/total life ensured to the product).

product and in default LCA calculations excluding such contribution. If only net differences in losses/waste of packaged product compared to a common reference are accounted, default loss rates in Annex B may be applied as an estimate of the base loss/waste of the reference alternative<sup>60</sup>, although the use of packaging-specific values should be preferred. Regardless of the chosen approach, all applied values and assumptions shall be adequately documents and justified, as well as submitted to the verification process. This is required to also ensure proper use of data and results in other studies and applications.

#### 4.2.4 Use stage

The *Use* stage describes how the product is expected to be used by the end user (e.g. the consumer). It begins at the moment the end user starts using the product and ends when the used product leaves its place of use and enters the *End of Life* stage (e.g. it is collected for recycling, disposal or other treatments). Collection and transport of the product to relevant end of life options is excluded from the *Use* stage, while it is part of the *End of Life* stage. Similarly, collection, transport and management of any loss/waste of the product in scope occurring during the *Use* stage, such as food waste packaging<sup>61</sup> or the product left at its end of use, is excluded from the *Use* stage and shall be part of the *End of Life* stage of the product.

The *Use* stage includes all activities and products that are needed for a proper use of the product (i.e. such that the provision of its original function is kept throughout the entire product lifetime; see Figure 4). Examples of use-stage activities and input or output flows that shall be included, as far as relevant, in the *Use* stage are:

- Refrigeration at the location of use; if required by the product in scope (e.g. food), - or in comparative LCA studies (conducted under a specific PEFCR) addressing packaging of products requiring refrigeration and where the compared alternatives involve substantially different refrigeration requirements for the packaged product<sup>62</sup>.
- Microwaving of ready-to-prepare packaged food (in case of relevant differences in time and/or power requirements among any compared packaging solutions under a specific PEFCR).
- Preparation for consumption by the end user (e.g. provision of tap water and wastewater treatment when cooking food as the product in scope; manufacturing, distribution and waste management of paper filters for coffee making<sup>63</sup>).
- Direct resource and product consumption during use (e.g. detergent, energy and water used to run a washing machine).
- Manufacturing, distribution and end-of-life management (i.e. life cycle) of materials needed for maintenance, repair or refurbishment of the product during the *Use* stage (e.g. spare parts needed to repair the product; production and end-of-life management of coolant used to compensate for its losses during use).
- Application of agricultural mulching film to soil, or application of other agricultural plastic products for the intended use (e.g. greenhouse covering or bale/silage wrapping).

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<sup>60</sup> The base loss/waste rate of the reference alternative is required to calculate the total loss/waste of packaging that shall be considered, for all compared alternatives, in default LCA calculations (even when applying a “differential” approach for additional LCA calculations accounting for losses/waste of packaged product). In such additional calculations, the loss/waste of packaged product of the reference alternative shall be set to zero, while for the other assessed alternatives only the net additional or avoided loss/waste compared to such a reference shall be considered.

<sup>61</sup> And related food content when included in calculations, see further in this section.

<sup>62</sup> Due to, e.g., substantially different product lives.

<sup>63</sup> Note that the end of life of paper filters used for coffee making shall be part of the *Use* stage (when coffee is the product in scope). Conversely, the end of life of coffee capsules, residues for coffee making and packaging of ground coffee belong to the *End of Life* stage.

- Application of insulation materials to buildings (especially if relevant differences exist among any alternatives possibly compared under a specific PEFCR).
- Use of plastic components (and related connection parts) in vehicles (e.g. fuel consumption and related emissions), in case alternatives with substantially different masses are compared under a specific PEFCR.
- Use of plastic components in any product where they substantially affect use-stage burdens (e.g. consumption of electricity or other consumables in electronic devices) or product performance during use, and/or relevant differences exist for these aspects among any alternatives compared under a specific PEFCR.
- Direct release of additives, micro-plastics or any other (degradation) compounds to air, water or soil (limited to any release taking place during the *Use* stage, not from the product after use).

The waste of ancillary products used during the *Use* Stage shall be included in the modelling of the *Use* Stage itself (accounting for the burdens from waste collection, treatment and/or disposal). The Circular Footprint Formula (Section 4.4.10.2) shall be applied to such waste.

The loss of product(s) in scope taking place during the *Use* Stage, shall also be included in the modelling of such stage, accounting for the related *Use* stage burdens until the point of loss generation. Collection, treatment and/or disposal of product loss shall be accounted in the End of Life stage, while upstream burdens from raw material extraction, manufacturing and distribution/storage of product loss shall be accounted in the relevant stages (i.e. *Raw Material Acquisition and Pre-processing, Manufacturing and Distribution*). Default loss rates per type of product at consumer level are provided in Annex B, and shall be used in case no specific and more representative information is available. The Circular Footprint Formula (Section 4.4.10.2) shall again be applied to such waste.

For LCA studies of packaging products (e.g. food packaging), the same rules provided for the *Distribution* stage apply regarding the handling of losses and waste of packaging and of the respective product content. This means that only losses/waste of packaging shall be accounted, by default, in LCA calculations related to the *Use* stage. However, if relevant differences exist among loss/waste rates when alternative packaging solutions are compared under a specific PEFCR, LCA results shall be additionally calculated also including the contribution of the life cycle<sup>64</sup> of losses/waste of packaged product (or at least of their net differences compared to a common reference alternative). Such additional results shall be presented and discussed separately. Further details on the related calculations are provided in Section 4.2.3 with reference to the *Distribution* stage.

In some cases, specific products are needed to allow a proper use of the product in scope, which then become physically integrated with it: in this case, the waste treatment of such products belongs to the *End of Life* stage of the product in scope. For example, when the product in scope is a detergent, the wastewater treatment of the water used to fulfil the function of the detergent belongs to the *End of Life* stage.

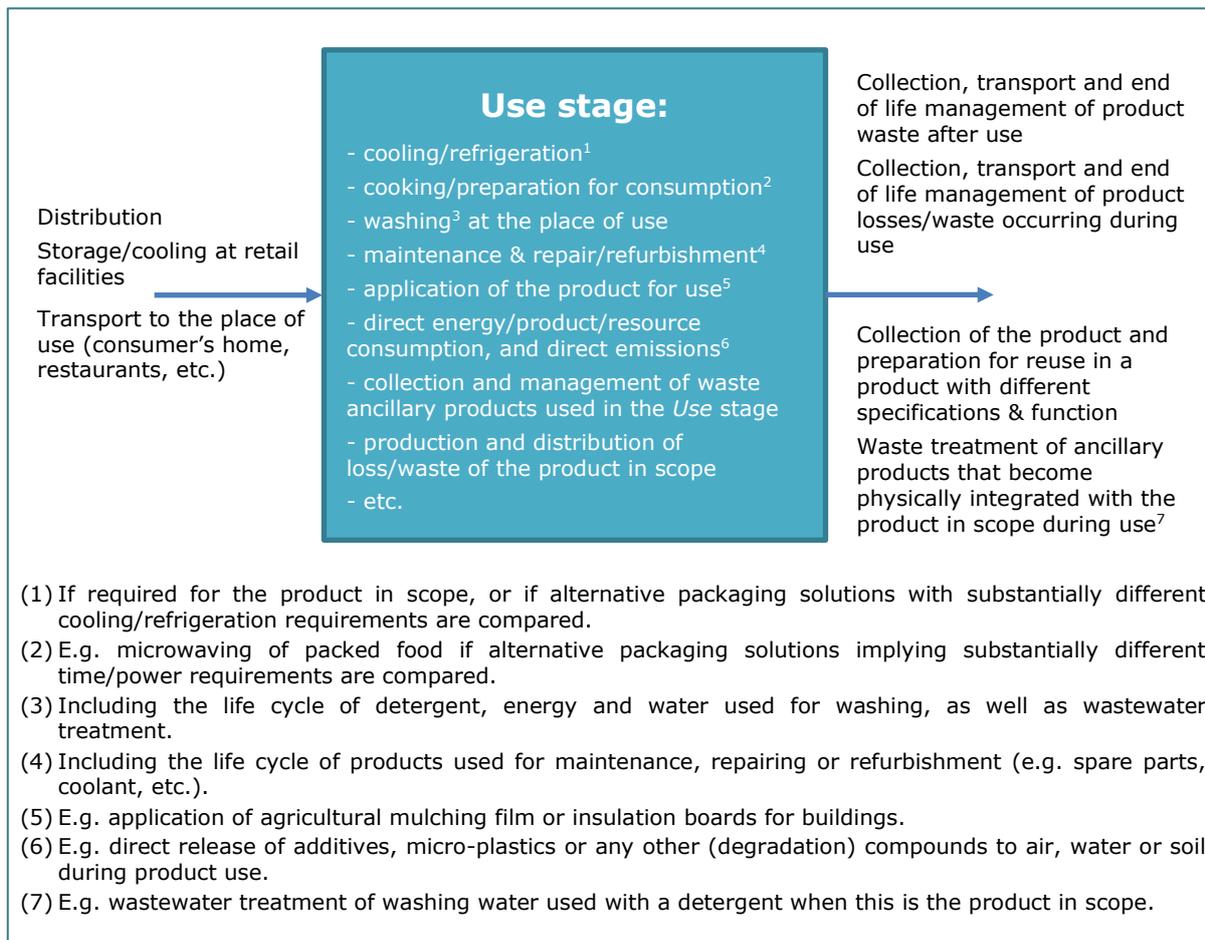
The following processes are also excluded from the *Use* stage:

- If a product is reused, the processes needed to collect the product and make it ready for the new use cycle (e.g. collection and cleaning of reusable bottles; see also Section 4.4.11). These processes are included in the *End of Life* stage (and modelled through the CFF) if the product is reused into a product with different specifications (i.e. providing a different function; see Section 4.4.11 for further details). If the product lifetime is extended into a product with original product specifications (providing the same function) these processes shall be included in the relevant upstream life cycle stages (e.g. *Distribution* for bottles collection and *Manufacturing* for cleaning) appropriately calculating the respective reference flow (see Section 4.4.11 for details).

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<sup>64</sup> Including Raw Material Acquisition and Pre-Processing, Manufacturing, Distribution and End of Life.

- Transport from retail to the place of use (e.g. consumer home) shall be excluded from the *Use* stage and shall be included in the *Distribution* stage.



**Figure 4.** Examples of processes and activities included and excluded from the *Use* stage.

The use scenario also needs to reflect whether or not the use of the analysed product(s) might lead to changes in the system(s) in which they are used, i.e. whether a system-system or part-system relationship<sup>65</sup> exists. For example, some energy-using products might affect the energy needed for heating/cooling in a building (system-system), while the weight of a car battery or of a car panel might affect the fuel consumption of the car (part-system). More in general, any plastic part used in mobile applications may substantially affect the total mass of such applications and hence the related energy (e.g. fuel) consumption. Similarly, the use of a given (plastic) material for specific components in electronic applications may affect its energy consumption during use. Inputs, outputs or activities of such related systems, which are substantially affected by the use of the analysed product, shall be included in the *Use* stage, especially if relevant differences exist among any product alternatives possibly compared under a specific PEFCR.

The following sources of technical information should be taken into account to define the use scenario (non-exhaustive list):

- Published international standards that specify guidance and requirements for the development of scenarios for the *Use* stage and scenarios for (i.e. estimation of) the service life of the product;

<sup>65</sup> Further details are available in EC-JRC (2010).

- Published national guidelines for the development of scenarios for the *Use* stage and scenarios for (i.e. estimation of) the service life of the product;
- Published industry guidelines for the development of scenarios for the *Use* stage and scenarios for (i.e. estimation of) the service life of the product;
- Market surveys or other market data.

NOTE: The method recommended by the manufacturer to be applied in the *Use* stage (e.g. cooking in an oven at a specified temperature for a specified time) should be used (if available) to provide a basis to determine the *Use* stage scenario of a product. The actual usage pattern may, however, differ from the one(s) recommended and should hence be applied if this information is available and documented.

Documentation of methods and assumptions shall be provided. All relevant assumptions for the *Use* stage shall be documented, and their relevance for the results or any conclusions shall be explicitly addressed in the results interpretation.

Technical specifications to model the *Use* stage are provided in Section 4.4.9.

The *Use* stage shall be excluded for intermediate products (e.g. polymers or unspecified plastic parts).

#### **4.2.5 End of Life (including product recovery and recycling)**

The *End of Life* stage begins when the product in scope (and its packaging) is discarded by the user and ends when the product is returned to nature as an emission (e.g. from incineration or landfilling) or enters another product's life cycle (i.e. as a recycled material input).

In general, the *End of Life* stage includes handling (collection, transport and treatment) of the waste of the product in scope after use and of its primary packaging, if any. Other waste streams (different from the product in scope) generated during manufacturing, distribution, retail, the use stage or after use shall be included in the life cycle of the product and modelled at the life cycle stage where they occur.

All end of life options and processes applied to the product in scope shall be considered in the LCA study, through suitable end of life scenarios, as detailed in Section 4.4.10.1. For innovative or emerging products (or products relying on innovative or emerging materials), potentially applicable End of Life options shall be considered.

Examples of end of life processes that shall be included, where applicable, in the *End of Life* stage include:

- Collection and transport of the product in scope and its packaging to end of life treatment/recovery facilities or disposal sites;
- Dismantling of components;
- Operations carried out at transfer stations;
- Operations carried out at material recovery or sorting facilities (e.g. bale opening, separation from impurities or other co-collected recyclable materials, sorting into homogeneous polymer streams and colours, etc.)<sup>66</sup>;
- Conversion into recycled material or feedstock via mechanical recycling<sup>67</sup>, chemical recycling (e.g. depolymerisation), or other forms of combined recycling (e.g. mechanical recycling followed by dissolution);

<sup>66</sup> Part of these operations (e.g. separation into homogeneous polymer streams and/or colours of pre-sorted plastic waste) may also be carried out directly at recycling facilities, which shall be included as well with the respective operations.

<sup>67</sup> Including all relevant operations carried out during the recycling process, such as possible sorting into homogeneous colours, grinding/shredding, cleaning/washing, flotation/water-based density separation, performance improvement, solid state polymerisation, blending with additives or other substances, granulation/extrusion, etc.

- Composting or other biological treatment options (e.g. anaerobic digestion) and on-land application of resulting organic material;
- Incineration and disposal of process residues (e.g. bottom ash);
- Landfilling (including landfill emissions, operation and maintenance);
- In-situ (bio)-degradation (e.g. biodegradation on/into the soil of agricultural mulching film);
- Wastewater treatment of products used dissolved in or with water (e.g. detergents, shower gels, etc.);
- Any waste mismanagement option (e.g. open burning, unsanitary landfilling) and product littering<sup>68</sup>, where relevant for the geography in scope.

For intermediate products, the *End of Life* stage of the product in scope shall be excluded.

The end of life of the product in scope and of all waste streams occurring during the *Manufacturing*, *Distribution* and *Use* stages shall be modelled using the Circular Footprint Formula and the requirements provided in Section 4.4.10.2.

### 4.3 Nomenclature for the Life Cycle Inventory

Life Cycle Inventory data applied in the study shall be compliant with the requirements for “EF-compliant data sets”<sup>69</sup>:

- For the elementary flows, the nomenclature shall be aligned with the most recent version of the Environmental Footprint (EF) reference package (currently 3.0) available on the EF developer’s page at the following link: <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>. Details to fulfil this requirement are available at: <http://eplca.jrc.ec.europa.eu/uploads/MANPROJPR-ILCD-Handbook-Nomenclature-and-other-conventions-first-edition-ISBN-finv1.0-E.pdf>.
- For the process datasets, product flows and waste flows, the nomenclature shall be compliant with the “ILCD Handbook – Nomenclature and other conventions” (available at: <http://eplca.jrc.ec.europa.eu/repository/EF>).

It shall also be ensured that existing process datasets applied in the inventory<sup>70</sup>, and the latest EF LCIA methods and characterisation factors, are implemented consistently with the latest EF reference package in the LCA software that may be used in the study.

### 4.4 Modelling requirements

This section provides detailed guidance and requirements or recommendations on how to model specific life cycle stages, processes or other aspects of the product life cycle, in order to compile the Life Cycle Inventory. The following stages, processes or modelling aspects are specifically addressed:

- Fossil-based feedstock supply;
- Use of plastic waste as a feedstock;
- Agricultural production;

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<sup>68</sup> Littering does not represent an intended End of Life option for (plastic) products, but rather a mismanagement practice, which similarly to accidents is normally not considered in LCA. However, due to the (current) relevance of this environmental issue for plastic products, the *End of Life* stage shall also account for product littering, with the respective burdens and impacts (as far as suitable methods are developed to address these). The estimated quantity of product ending up as littered into the environment at End of Life (per functional unit) shall also be reported as “additional environmental information” (as discussed in Section 3.2.5).

<sup>69</sup> Available at [https://eplca.jrc.ec.europa.eu/permalink/Guide\\_EF\\_DATA.pdf](https://eplca.jrc.ec.europa.eu/permalink/Guide_EF_DATA.pdf)

<sup>70</sup> Whether they are EF-compliant or ILCD-Entry Level compliant.

- Use of bio-based waste or by-products as a feedstock;
- Use of CO<sub>2</sub> from gaseous effluents as a feedstock;
- Handling of products based on maturing or emerging technologies;
- Transport and logistics;
- Storage at distribution centres or retail;
- Use stage;
- End of Life modelling;
- Extended product lifetime;
- Electricity use;
- Capital goods (infrastructures and equipment);
- Sampling procedure;
- Greenhouse Gas (GHG) emissions and removals;
- Offsets (GHG emissions).

Other aspects relevant to the compilation of the Life Cycle Inventory are also addressed in Sections 4.5, 4.6 and 4.7, including:

- Handling of multi-functional processes;
- Data collection requirements;
- Cut-off;
- Data quality assessment and quality requirements.

#### **4.4.1 Fossil-based feedstock supply**

This section addresses relevant activities and aspects that shall be considered, as a general rule, when modelling the supply of fossil-based feedstock sources for polymer production (i.e. mainly crude oil and naphtha, although also natural gas is used). Usually, however, such aspects are fully considered in available background datasets representing fossil-based feedstock supply, and do not directly concern the practitioner conducting a LCA study on plastic products following this method, except for more rare cases where company-specific data can be directly accessed for these activities from specific suppliers in the value chain of the product in scope<sup>71</sup>. On the other hand, LCI data providers and dataset developers shall ensure that all environmentally relevant activities and aspects related to fossil-based feedstock supply are appropriately taken into account and modelled based on up to date and representative data when developing new datasets (or updating existing ones), considering the requirements and recommendations provided below.

Relevant activities that shall be considered, with the related environmental burdens, in the modelling of fossil-based feedstock supply include:

- Oil exploration, including any related drilling operations (especially for most recent oil sources, e.g. deep-water ones);
- Well drilling;
- Oil extraction/production and possible pre-processing;

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<sup>71</sup> According to the data collection requirements reported in Section 4.6, activities related to feedstock supply can be modelled based on secondary life cycle inventory data, to be selected according to the hierarchy reported in Section 4.6.3. This applies to both final and intermediate products (e.g. polymers).

- Long-distance oil transport via pipeline and/or tanker vessels (including emissions into the environment due to any structural oil losses and compensation for these; see below);
- Regional distribution to final users (e.g. via pipeline);
- Oil refining (e.g. production of naphtha or other relevant intermediate feedstocks);
- Naphtha transport to downstream users (e.g. cracking facilities), if applicable.

Crude oil supply shall reflect, as far as possible, the average supply mix to refineries in the relevant geography for the specific supply chain investigated. The mix is intended in terms of country of origin and corresponding type of oil sources (including conventional and unconventional sources such as oil sands, shale oil, etc.) and extraction/processing technologies.

Oil exploration activities and related drilling operations should also be taken into account in the burdens associated with crude oil supply. This should be especially the case for most recent (off-shore) oil fields, which require ongoing (and generally deep-water) exploration and drilling activities compared to sources placed in locations established from decades (Unnasch et al., 2009).

LCA studies typically look at normal (average) production conditions, excluding burdens and potential impacts from accidents, disasters or conflicts, such as those from accidental oil spills and oil fires during extraction activities<sup>72</sup>, or from military operations performed to protect oil supply. However, specific accidental emissions that can be considered a structural property of the supply chain are normally taken into account. Therefore, structural losses during oil transport should be accounted for (as also normally done in most existing datasets representing crude oil supply). These should be modelled in terms of direct oil emissions into the relevant environmental compartment(s) and of increased oil production to compensate for losses and provide the intended oil output. It must be noted, however, that biophysical and toxicological impacts of oil directly emitted to seawater on animals or other (marine) ecosystems are not captured within traditional LCA impact categories (including the default categories adopted in this method). They may be reported as “additional environmental information” (e.g. as part of biodiversity impacts). Toxicological impacts on freshwater ecosystems are instead captured in the *Ecotoxicity – freshwater* impact category considered in this method after the release of the 3.0 version of the EF reference package (i.e. the most recent currently available and to be applied in LCA studies conforming to this method). The latter indeed includes characterisation factors for unspecified oil emissions to seawater (and other environmental compartments), calculated as the average of values related to several refinery products<sup>73</sup>.

Other potentially relevant emissions directly to seawater may occur from (off-shore) oil exploration, drilling, extraction and transport activities. These include, for instance, emissions of drilling mud to seabed (exploration and drilling activities), release of metals from oxidation of sacrificial anodes on pipelines used for oil transport, as well as emissions from the use of auxiliaries in large quantities offshore (e.g. particles of coatings required to protect structures operating in corrosive conditions). While such emissions are frequently excluded from life cycle inventories related to crude oil supply, their relevance in terms of quantity and resulting potential impacts should be checked when developing new datasets or updating existing ones, and their contribution shall then be included accordingly.

Land transformation and occupation burdens shall be accounted for land-based oil sources (e.g. oil sands) considering suitable elementary flows for both previous and

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<sup>72</sup> Similarly to the burdens of improper production conditions or cultivation practices, such as misuse of pesticides and fertilisers in agriculture.

<sup>73</sup> Note, however, that characterisation factors for unspecified oil emissions are lower compared to those of a large portion of substances covered in the *Ecotoxicity – freshwater* impact category (i.e. nearly 70% of them).

current land use. While land-based sources also normally generate impacts on landscape (e.g. visual/aesthetic impacts from mining), these are typically not captured in LCA (nor in the default impact categories considered in this method), and no specific flows are generally inventoried in this respect.

It is also important to ensure that up to date and representative data appropriately reflecting the situation at the time the LCA study is developed are used to model the processes and activities listed above. This is particularly relevant, for instance, for energy consumption associated with oil (and gas) extraction activities, which should take into account the age of the fields where extraction takes place, since energy requirements for extraction generally increase over time (due to e.g. larger amounts of water to be re-injected in the fields themselves). Data related to the initial energy demand for extraction from (oil) fields established at the end of the last century would hence not be representative of extraction activities carried out today on the same older fields.

Providing more detailed guidance on the modelling of individual processes and activities related to fossil-based feedstock supply is beyond the scope of this method. However, the provisions reported above should ensure that all environmentally relevant processes, activities and aspects related to this part of the life cycle are properly taken into account when developing or updating life cycle inventories in this respect.

#### **4.4.2 Use of plastic waste as a feedstock**

The use of post-consumer plastic waste as a feedstock for the production of new polymers, and subsequently of specific plastic products, is a situation of recycling. At present, this mostly takes place via mechanical recycling processes, where mixed post-consumer plastic waste from municipal or industrial collection (or specific plastic waste streams segregated directly at source, e.g. from take-back systems) are typically sorted into specific polymer categories (and then possibly by colour), and subsequently processed into secondary polymer flakes or granulate for use in the manufacturing of new products. In emerging chemical or chemo-physical recycling routes, plastic waste is instead converted into either new dissolved polymers (e.g. via dissolution), monomers (depolymerisation or chemolysis/solvolysis), or into a liquid (oil-like) or gaseous feedstock (via pyrolysis or gasification) to be used as an input to conventional refinery or cracking processes providing chemicals or intermediates also used in polymer production.

As in any recycling/material recovery situation, the Circular Footprint Formula (CFF) shall be applied to model the use of post-consumer plastic waste for the production of plastic polymers or products. The formula is described in Section 4.4.10.2, and should be applied at the most proximate (upstream) point of substitution in the supply chain where recycled material is proven to replace an equivalent virgin material as an input to a same (or very similar) process, or where a product incorporating recycled material replaces an equivalent product (mostly produced) from virgin material. In the case of plastic products based on mechanically recycled material, the point of substitution can be often identified at the level of recycled polymer granulate replacing virgin polymer resin of the same material as an input to product manufacturing. However, other points of substitution downstream in the supply chain may be identified in specific situations. Further details on how to apply the CFF are available in Section 4.4.10.2.

#### **4.4.3 Agricultural production**

Agricultural production is an essential part of the life cycle of food, feed and other bio-based products, including bio-polymers and bioplastic products. Activities related to agricultural production (as well as to plantation, forestry and other open production systems) are specific cases where more detailed modelling guidance is needed to ensure a sufficient level of representativeness and reproducibility. This is because, in contrast to industrial production systems, emissions to the environment and specific inputs (e.g. net water consumption) from such activities cannot be practically measured, but need to be modelled or calculated, thus requiring specific modelling provisions.

The modelling guidelines reported in this section shall be followed when developing life cycle inventories of agricultural production processes.

#### **4.4.3.1 Activities, inputs and emissions**

The following activities, inputs and emissions shall be considered, where applicable, when modelling agricultural production, unless their exclusion is allowed based on the cut-off criteria (Section 4.6.4):

- Seeds and/or seedlings production, and application (seeding and/or bedding);
- Fertilisers (synthetic and organic) production and application;
- Peat extraction, processing and application;
- Lime mining, processing and application;
- Pesticides production and application;
- Mulch film production, application, and fate after use;
- Transport of agricultural inputs (including those listed above) to the field;
- Irrigation (including water consumption and any related energy input);
- Use of agricultural machinery (and associated fuel consumption and emissions) for soil treatment and other field operations beyond those associated with the application of the inputs listed above (e.g. harvesting);
- Input of N, P and other nutrients from crop residues that are left on or applied to the field;
- Field emissions of N and P (compounds) from fertilisers application;
- CO<sub>2</sub> emissions from application of lime, urea or urea-compounds;
- Emissions from pesticides application;
- Emissions of heavy metals (including the possible contribution from fertilisers and pesticides);
- Emissions from burning of residues (before or after harvesting);
- Drying and storage of products (shall always be included, unless its exclusion is clearly justified based on the cut-off criteria).

All biogenic carbon removals and emissions shall be modelled separately, including CO<sub>2</sub> uptakes from crops/plants during growth (see Section 4.4.15 for more details on the modelling). However, note that characterisation factors for biogenic CO<sub>2</sub> uptakes and emissions are set to zero within the default EF impact assessment method for the Climate Change impact category (which shall be applied also by users of this method; Section 3.2.4).

Relevant parameters and information that should be known (collected on-site or calculated) for the modelling of activities and emissions related to fertilisers use are the specific type of fertilisers applied (e.g. in terms of origin –mineral or organic–, type of compound(s) and main constituents), the application rate (or dose, e.g. as kg applied per hectare of cultivated land) and the respective nutrient (N, P, K) content. For peat, the respective C/N ratio is relevant beyond the application rate, while for lime the type applied and its carbonate (CaCO<sub>3</sub>) content should be known in addition to the applied dose. Similarly to fertilisers, the application rate (kg residue/ha) and nutrient content are both relevant for crop residues left on or applied to the field, while for pesticides the composition in terms of active substance(s) is especially relevant, along with the application rate.

For seed material and seedlings, relevant parameters include the application rate or the planting density (to be possibly calculated based on field configuration), while for

mulching film the share of cultivated land actually covered with mulch should be known, along with the thickness and density of the film. The fate of the film itself after use and related end-of-life parameters are also relevant (e.g. the collection rate and the shares of applied end-of-life alternatives). In the case of irrigation, the specific water consumption per hectare should be typically acquired or calculated. For machinery use in agricultural operations, the specific type of operation and machinery applied should be known, as well as the respective duration of use, i.e. the time needed to perform the intended operation (e.g. hours per hectare).

Further details on the modelling of specific activities and emissions, or on the handling of specific aspects (e.g. multi-functional processes), are provided in the following subsections (4.4.3.2 – 4.4.3.10).

#### **4.4.3.2 Handling multi-functional processes**

The rules described in the LEAP Guideline (FAO, 2015) shall be followed: *Environmental performance of animal feeds supply chains – Guidelines for quantification* (pages 36-43), available at: <http://www.fao.org/partnerships/leap/publications/en/>.

#### **4.4.3.3 Crop type-specific and country-, region- or climate-specific data**

Crop type specific and country-region-or-climate specific data should be used for the following parameters (and expressed per hectare and per year):

- Yield;
- Net water use by crops (i.e. loss through evaporation and transpiration);
- Land use;
- Land use change;
- Applied synthetic or organic fertilisers (N, P, K amount and specific compounds applied, others as applicable);
- Pesticides amount applied (per active ingredient).

If relevant for the scope of the study, data used for these parameters shall also be representative of any specific crop management schemes applied.

Country- or region-specific averages should be preferred over climate-specific averages. Marginal crops should be considered only if it is known that they are used as a feedstock for the product in scope.

#### **4.4.3.4 Averaging data**

Cultivation data shall be collected over a period of time sufficient to develop an average life cycle inventory of the inputs and outputs of cultivation activities, where fluctuations due to seasonal differences are offset (exceptions are addressed immediately below). This shall be performed as described in the LEAP guidelines (FAO, 2015), as set out below (except for the last point on land-use, which is added in this guide):

- For annual crops, an assessment period of at least three years shall be used (to level out differences in crop yields related to fluctuations in growing conditions over the years such as climate, pests and diseases, etc.). Where data covering a three-year period are not available, due for instance to starting up a new production system (e.g. a new greenhouse, newly cleared land, or shift to another crop), the assessment may be conducted over a shorter period, but this shall not be less than 1 year. Crops or plants grown in greenhouses shall be considered as annual crops/plants, unless the cultivation cycle is significantly shorter than one year and another crop is cultivated consecutively within that year. Tomatoes, peppers and other crops that are cultivated and harvested over a longer period throughout the year are considered as annual crops. Averaging data (e.g. emissions) over three years may best be done by first

gathering annual data and calculating the life cycle inventory per year, and then determining the three-year average.

- For perennial plants (including entire plants and edible portions of perennial plants) a steady state situation (i.e. where all development stages are proportionally represented in the studied time period) shall be assumed and a three-year period shall be used to estimate the inputs and outputs<sup>74</sup>.
- Where the different stages in the cultivation cycle are known to be disproportional in the studied time period (e.g. a high share of cultivated area is just renewed and do not provide yield), a correction shall be made by adjusting the crop areas allocated to the different development stages in proportion to the crop areas expected in a theoretical steady state. The application of such correction shall be justified and recorded. A life cycle inventory of perennial plants and crops shall not be performed until the production system actually yields output.
- For crops that are grown and harvested in less than one year (e.g. lettuce produced in 2 to 4 months) data shall be gathered in relation to the specific time period for production of a single crop, from at least three recent consecutive cycles.
- To determine the land use duration, periods between different crop-cycles are to be accounted for as well, by assigning the time period between the harvest of the preceding crop and the sowing or planting of the crop that is used in the analysed product system, to such product system.

#### **4.4.3.5 Fertilisers**

Fertiliser (and manure) emissions shall be differentiated per fertiliser type, and cover as a minimum:

- NH<sub>3</sub>, to air (from N-fertilisers application);
- N<sub>2</sub>O, to air, including both direct and indirect emissions, with indirect emissions coming from NH<sub>3</sub> and NO<sub>x</sub> released to air and from NO<sub>3</sub><sup>-</sup> leaching/runoff) (from N-fertilisers application);
- CO<sub>2</sub>, to air (from lime, urea and urea-compounds application);
- NO<sub>3</sub>, to *water unspecified* (leaching from N-fertilisers application);
- PO<sub>4</sub>, to *water unspecified* or *freshwater* (leaching and run-off of soluble phosphate from P-fertilisers application);
- P, to *water unspecified* or *freshwater* (runoff of soil particles containing phosphorous, from P-fertilisers application).

Emission compartments specified above (in italics) for N and P releases due to leaching and runoff are defined taking into account characterisation factors available in the default impact assessment methods for the categories *Eutrophication – freshwater* and *Eutrophication – marine*, and are based on the following considerations.

The default impact assessment model for freshwater eutrophication (*Eutrophication – freshwater*) can start: (i) either when P or P compounds leave the agricultural field/soil (after leaching or run off) and are emitted in the (first) waterbody (i.e. starting from emissions occurring directly to water after leaching/runoff from agricultural soil has taken place), or (ii) from fertilisers or manure application on agricultural field (i.e. from

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<sup>74</sup> The underlying assumption for cradle-to-gate life cycle inventories of perennial agricultural products is that the inputs and outputs of the cultivation are in a 'steady state', which means that all development stages of perennial crops (with different quantities of inputs and outputs) shall be proportionally represented in the time period of cultivation that is studied. This approach gives the advantage that inputs and outputs of a relatively short period can be used for the calculation of the cradle-to-gate life cycle inventory of a perennial crop product. Studying all development stages of a perennial agricultural crop can have a lifespan of 30 years and more (e.g. in case of fruit and nut trees).

emissions occurring directly to agricultural soil after fertilisers/manure are applied on it). However, within LCI modelling, the agricultural field (soil) is typically seen as belonging to the technosphere and thus included in the LCI model (i.e. direct emissions to agricultural soil are modelled until they reach the first real environmental compartment outside agricultural soil). This aligns with approach (i), where the impact assessment model starts after leaching or run-off has occurred within agricultural soil, i.e. when P leaves the agricultural field/soil. Therefore, P emissions should be normally inventoried as the amount of P or P compounds emitted to water after leaching or run-off has taken place, and the emission compartment 'water' shall be used (i.e. 'water, unspecified' for emissions from leaching<sup>75</sup>, and 'freshwater' for emissions from runoff). When this amount is not available, the emissions may be inventoried as the amount of P directly applied on the agricultural field (through fertilisers or manure) and the emission compartment 'soil' (e.g. 'agricultural soil') shall be used. In this case, run-off and leaching from soil to water are modelled as part of the default impact assessment model and included in the provided characterisation factors for emissions to 'soil'. This approach and resulting factors come however with a higher uncertainty, and is less specific for the actual field management, climate setting, soil conditions, and other relevant site-specific aspects.

The default impact assessment model for marine eutrophication (*Eutrophication – marine*) starts after N leaves the field (agricultural soil). Therefore, N emissions from application of fertilisers shall not be modelled as bare emissions to (agricultural) soil. Conversely, the amount of nitrogen compounds released in the different air and water end compartments (per amount of fertilisers applied on the field) shall be modelled within the inventory.

Nitrogen emissions shall be calculated based on nitrogen applications by the farmer on the field excluding any external sources (e.g. rain deposition), by balancing the overall N input to the field and output with harvest plus storage-change.

To avoid strong inconsistencies among different LCA studies and PEFCRs possibly developed based on this method, a number of emission factors are fixed, by following a simplified approach. For nitrogen-based fertilisers, the Tier 1 emission factors from IPCC (2006; Chapter 11) should be used, as presented in Table 5<sup>76</sup>. In case better data are available, a more comprehensive nitrogen field model may be used in the LCA study, provided that: (i) it covers at least the emissions requested above (Table 5); (ii) nitrogen shall be balanced in inputs and outputs and (iii) it shall be described in a transparent way.

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<sup>75</sup> No specific characterisation factors are provided for emissions to 'groundwater' in the default impact assessment method for freshwater eutrophication.

<sup>76</sup> Note that accuracy of the values provided is scarce, and they shall not be used to compare different types of fertilisers. They are also unsuitable to identify or differentiate better and worse nitrogen management practices of the field.

**Table 5.** Tier 1 emission factors of IPCC for nitrogen emissions from application of fertilisers (adapted from IPCC, 2006). Note that these values shall not be used to compare different types of fertilisers.

Emission	Compartment	Emission factor to be applied
N <sub>2</sub> O (synthetic fertilisers and manure; direct and indirect)	Air	<b>0.022 kg N<sub>2</sub>O/ kg N in the fertiliser applied</b>
NH <sub>3</sub> - Urea (synthetic fertiliser)	Air	kg NH <sub>3</sub> = kg N * FracGASF <sub>U</sub> <sup>(1)</sup> = 1*0.15*(17/14) = <b>0.18 kg NH<sub>3</sub>/ kg N in the fertiliser applied</b>
NH <sub>3</sub> - Ammonium nitrate (synthetic fertiliser)	Air	kg NH <sub>3</sub> = kg N * FracGASF <sub>AN</sub> <sup>(2)</sup> = 1*0.1*(17/14) = <b>0.12 kg NH<sub>3</sub>/ kg N in the fertiliser applied</b>
NH <sub>3</sub> - others (other synthetic fertilisers)	Air	kg NH <sub>3</sub> = kg N * FracGASF <sub>F</sub> <sup>(3)</sup> = 1*0.02*(17/14) = <b>0.024 kg NH<sub>3</sub>/ kg N in the fertiliser applied</b>
NH <sub>3</sub> (manure)	Air	kg NH <sub>3</sub> = kg N * FracGASF <sub>M</sub> <sup>(4)</sup> = 1*0.2*(17/14) = <b>0.24 kg NH<sub>3</sub>/ kg N in the manure applied</b>
NO <sub>3</sub> <sup>-</sup> (synthetic fertilisers and manure) <sup>(5)</sup>	Water	To be calculated based on the NO <sub>3</sub> -N base loss according with Equation 1 below

<sup>(1)</sup> FracGASF<sub>U</sub>: fraction of nitrogen contained in urea that is emitted as ammonia to air.

<sup>(2)</sup> FracGASF<sub>AN</sub>: fraction of nitrogen contained in ammonium nitrate that is emitted as ammonia to air.

<sup>(3)</sup> FracGASF<sub>F</sub>: fraction of nitrogen contained in other types of synthetic fertilisers that is emitted as ammonia to air.

<sup>(4)</sup> FracGASF<sub>M</sub>: fraction of nitrogen contained in manure that is emitted as ammonia to air.

<sup>(5)</sup> Total NO<sub>3</sub>-N emission to groundwater via leaching (indicated in Equation 1 below as ΣNO<sub>3</sub>-N<sub>W</sub>).

As specified in Table 5, the total NO<sub>3</sub>-N emission to groundwater via leaching (ΣNO<sub>3</sub>-N<sub>W</sub>, which is considered a variable) shall be calculated as:

$$\Sigma NO_3-N_W = NO_{3,BL}^- + NO_3-N_{AW} \quad \text{[Equation 1]}$$

Where ΣNO<sub>3</sub>-N<sub>W</sub> is the total NO<sub>3</sub>-N emission to groundwater, NO<sub>3</sub><sup>-</sup>,BL is the NO<sub>3</sub>-N base loss from synthetic fertilisers and manure application (NO<sub>3</sub><sup>-</sup> base loss), and NO<sub>3</sub>-N<sub>ATW</sub> are the additional NO<sub>3</sub>-N emissions to groundwater.

The NO<sub>3</sub><sup>-</sup> base loss shall be calculated as reported in Table 6, while the additional NO<sub>3</sub>-N emissions to water is quantified using Equation 2:

$$NO_3-N_{ATW} = \Sigma N_f + N_{2,fix} - N_R - NH_{3,air} - N_{2O,air} - N_{2,air} - NO_{3,BL}^- \quad \text{[Equation 2]}$$

Where:

- ΣN<sub>f</sub> is the N input to the field with all fertilisers (synthetic and manure);
- N<sub>2,fix</sub> is the N<sub>2</sub> fixation by crop (Table 6);
- N<sub>R</sub> is the N-removal with the harvest;
- NH<sub>3,air</sub> is the total NH<sub>3</sub> emissions to air (calculated as the sum of all contributions reported in Table 5, as far as relevant to the analysed process);
- N<sub>2O,air</sub> is the N<sub>2</sub>O emissions to air (Table 5);
- N<sub>2,air</sub> is the N<sub>2</sub> emissions to air (Table 6);
- NO<sub>3</sub><sup>-</sup>,BL is the NO<sub>3</sub><sup>-</sup> base loss (Table 6).

If in certain low-input schemes the value for the additional NO<sub>3</sub>-N emissions to water (NO<sub>3</sub>-N<sub>ATW</sub>) is negative, the value is to be set to "0". Moreover, in such cases the absolute value of the calculated additional NO<sub>3</sub>-N emissions to water is to be inventoried as additional N-fertiliser input into the system, using the same combination of N-fertilisers as for the analysed crop. This last step serves to avoid fertility-depletion schemes by capturing the N-depletion by the analysed crop, which is assumed to lead to the need for additional fertiliser later on and to keep the same soil fertility level.

**Table 6.** Additional parameters required to quantify the total NO<sub>3</sub>-N emission to groundwater according to Equation 1 (adapted from IPCC, 2006).

Parameter	Compartment	Value to be applied
NO <sub>3</sub> <sup>-</sup> base loss (synthetic fertilisers and manure) (NO <sub>3</sub> <sup>-</sup> <sub>BL</sub> )	Water	kg NO <sub>3</sub> <sup>-</sup> = kg N * FracLEACH <sup>(1)</sup> = 1*0.1*(62/14) = 0.44 kg NO <sub>3</sub> <sup>-</sup> / kg N in the fertiliser or manure applied
N <sub>2</sub> -fixation by crop (N <sub>2,fix</sub> )	-	For crops with symbiotic N <sub>2</sub> -fixation: the fixed amount is assumed to be identical to the N-content in the harvested crop
N <sub>2,air</sub>	Air	0.09 kg N <sub>2</sub> / kg N in the fertiliser or manure applied

<sup>(1)</sup> FracLEACH: fraction of nitrogen contained in synthetic fertilisers and/or manure that is leached as nitrate to groundwater.

#### 4.4.3.6 Pesticides

Pesticide emissions shall be modelled as specific active ingredients. The default impact assessment model for the toxicity-related impact categories considered in this method (i.e. the USEtox model, Table 4) has a built-in multimedia fate model that simulates the fate of pesticides starting from the different possible (off-field) emission compartments (soil, air, freshwater). Therefore, default emission fractions are needed in the LCI modelling, to split the amount of pesticides applied on the field among the different environmental emission compartments (Rosenbaum et al., 2015). As default approach, the pesticides applied on the field shall be modelled as 90% emitted to the agricultural soil compartment, 9% emitted to air and 1% emitted to freshwater (based on expert judgement due to current limitations<sup>77,78</sup>). More specific data may be applied if available<sup>79</sup>, and shall be adequately justified and documented.

A robust and widely applicable pesticide emission model to assess the fate of pesticides applied on the field and to estimate the amount released in the different emission compartments is still missing today. The PestLCI model (Dijkman et al. 2012) or models derived from the OLCA-pest project might fill this gap in the future, but are currently still under testing. Note, however, that updated specific guidance on the modelling of pesticide emissions is currently being developed within the Agriculture Working Group of

<sup>77</sup> Several databases consider that the amount of pesticides applied to the field is 100% emitted to soil out of simplification (e.g. Agribalyse and Ecoinvent), while others model pesticide-specific and crop type- and application type-specific distributions to the different environmental media (e.g. GaBi). It is recognized that emissions to freshwater and air do occur. However, emission fractions vary substantially depending on the type of pesticide, the geographical location, the time of application and the application technique (ranging from 0% to 100%). Especially, the % emitted to freshwater can be strongly debated, although it seems that 1% indicates a reasonable average (see e.g. WUR-Alterra, 2016). On the other hand, emissions to groundwater can also be relevant for some pesticides and soil conditions, reaching 10-30% of applied active ingredients (Fantin et al., 2019).

<sup>78</sup> Please note that the prescribed emission fractions are temporary values until further work fills current gaps. The Agriculture modelling Working Group of the EF transition phase is developing such guidance for future use.

<sup>79</sup> Other environmental compartments may also be considered to ensure alignment between modelled LCI flows and CFs available in the default LCIA methods (e.g. for toxicity-related impact categories), especially in terms of fate modelled at the LCI level and by the applied LCIA models.

the Environmental Footprint initiative, and is expected to be available by the end of the ongoing EF transition phase (2019-2021).

#### **4.4.3.7 Heavy metal emissions**

Heavy metal emissions from field inputs shall be modelled as emissions to soil and/or from leaching or erosion (run-off) to water (*freshwater* for run-off emissions and '*water, unspecified*' for leaching emissions). Inventoried emissions to water shall specify the oxidation state of the metal (e.g. Cr<sup>+3</sup>, Cr<sup>+6</sup>), if the corresponding elementary flows are part of the most recent EF elementary flow list available at the time of the study (currently EF 3.0). As crops assimilate part of the heavy metal emissions occurring during their cultivation, clarification is needed on how to model crops that act as a sink. Two different modelling approaches are allowed, with a preference for the first option in the list below:

- The final fate of heavy metal emissions taken up by crops is considered within the system boundary: the inventory does account for final emissions of heavy metals taken up by crops in the environment and therefore shall also account for the uptake of heavy metals by the investigated crop and their subsequent fate until they are released back to the environment. For example, heavy metals taken up by agricultural crops cultivated for feed will mainly end up in animal digestion and then in manure applied back on the field, where the metals are released in the environment and their impacts captured by the impact assessment methods. In this case, the inventory of the agricultural production stage of the investigated crop shall account for the uptake of heavy metals by the crop itself and include their further release after manure application. A limited amount ends up in the animal (=sink), which may be neglected for simplification. It is acknowledged that it may be difficult to know the final fate of heavy metals after they are taken up by crops.
- The final fate of heavy metal emissions taken up by crops are not further considered within the system boundary: the inventory does not account for final emissions of heavy metals taken up by crops in the environment and therefore shall not account for the uptake of heavy metals by the investigated crop and their subsequent fate and release back to the environment. For example, heavy metals taken up by agricultural crops cultivated for human consumption end up in human digestion. Within this method human consumption is not modelled, the final fate is hence not further modelled and the plant acts as a heavy metal sink. Therefore, the uptake of heavy metals by the crop shall not be modelled, and no additional emissions shall be inventoried for the investigated crop.

#### **4.4.3.8 Rice cultivation**

Methane emissions from rice cultivation shall be included based on the calculation rules of IPCC (2006) (Volume 4, Chapter 5.5, page 44-53).

#### **4.4.3.9 Peat soils**

Drained peat soils shall include carbon dioxide emissions based on a model that relates the drainage levels to annual carbon oxidation. Default values reported in Table 7 should be applied to quantify CO<sub>2</sub> emissions from drained peat soils, unless more specific values are available and can be properly justified.

**Table 7.** Default values for CO<sub>2</sub> emissions from drained peat soils (in tonnes CO<sub>2</sub>/ha/year) (FAO, 2015).

Climate	Land Use			
	Forest land / Agroforestry	Cropland	Grassland	Extraction sites
Tropical	40	40	40	30
Subtropical	30	35	30	25
Temperate	20	25	20	15
Boreal	7	25	10	10

#### **4.4.3.10 Agricultural operations**

Unless it is clearly documented that agricultural operations (e.g. soil treatments, irrigation, harvesting, etc.) are carried out manually, they shall be considered to take place through agricultural machinery. The respective burdens shall be accounted in terms of total fuel consumption and resulting airborne emissions, or through specific background datasets quantifying the burdens of each specific field operation (possibly covering additional burdens such as the machinery life cycle and soil emissions from tyre abrasion).

Similarly, irrigation shall be modelled through the respective water consumption and the related energy/fuel demand (e.g. for pumping), or through a more comprehensive specific background dataset.

Transport to/from the field (e.g. of specific equipment) shall also be accounted for, where relevant.

#### **4.4.4 Use of bio-based waste or by-products as a feedstock**

A range of bio-based waste, by-products or residual materials may be used as a feedstock for bio-based polymers and plastic products. Some of these feedstock sources have already been explored, others may be in the future. They include, for instance, agricultural residues (e.g. wheat straw or corn stover), forestry products' processing residues or waste (e.g. sawmill wood chips, wood bark and sawdust), organic waste or by-products from industry (e.g. orange peels, shrimp shells, sugarcane bagasse, reclaimed starch from potato processing wastewater, residual fats/oils from animal or plant processing), the organic fraction of municipal waste, used cooking oils, and organic matter from (municipal) wastewater.

The appropriate Life Cycle Inventory modelling of these feedstock materials depends on whether they stem from:

- (a) Joint production/generation with other products (e.g. wheat straw, sugarcane bagasse, sawdust, reclaimed starch from potato processing, shrimp shells, residual fats/oils from industrial processing, etc.); or whether they are
- (b) An actual End of Life product (e.g. organic municipal waste, waste fats/oils after use, (most) organic matter in wastewater, etc.).

In the case of joint production, two situations are to be distinguished:

- a.1) The feedstock material has a positive economic value (market price) above 0 at the point where it occurs (i.e. excluding any storage, transport, additional processing, etc.): in this case it is a by-product and shall be modelled as any other co-product, i.e. applying the multi-functionality decision hierarchy reported in Section 4.5.

- a.2) The feedstock material has a market value of 0 or below at the point of arising: in this case it is a waste and its further handling, storage, processing, transport etc. shall be modelled until the first product is obtained that can, and in practice will, replace an alternative material (or energy carrier) that regularly is produced from primary resources. Substitution at the point of substitution and the Circular Footprint Formula shall then be applied as in any other recycling/material recovery situation (Section 4.4.10.2).

If the feedstock material is in contrast an end-of-life product (situation b), it equally shall be modelled until the point of substitution and the Circular Footprint Formula shall be applied.

The economic value (market price) considered in the analysis shall refer to the time period in which the LCA study is performed and to the specific geography where the feedstock is sourced (i.e. it shall refer to the case-specific situation in time and space).

#### **4.4.5 Use of CO<sub>2</sub> from gaseous effluents as a feedstock**

This section provides general methodological requirements on the modelling of the use of CO<sub>2</sub> as a feedstock for the production of plastic products via Carbon Capture and Utilisation (CCU) processes. Capture can take place from both point emission sources (e.g. power plants, cement kilns, ammonia production facilities, hydrogen production plants, etc.) or directly from air (Direct Air Capture – DAC). The focus of this section is on CO<sub>2</sub> captured from point emission sources.

Recent literature on relevant methodological aspects for CO<sub>2</sub> utilisation systems was considered (Dammer et al., 2018; Giegrich et al., 2018; Zimmermann et al., 2018; Von der Assen et al., 2014; Von der Assen et al., 2013), leading to the modelling requirements detailed below, and to the respective background methodological considerations reported in Annex G (Section G.1). Due to the innovative nature of CO<sub>2</sub>-based pathways for polymer production (and of CCU systems in general), the following requirements may be revised in the future, as far as CO<sub>2</sub>-based production routes and the market of captured CO<sub>2</sub> and of CO<sub>2</sub>-based polymers and plastic products possibly become more established. Providing more specific modelling requirements is not feasible at the current stage.

It is noted that, in general, the ultimate potential environmental impacts of CO<sub>2</sub>-based polymers and products will be affected, among others, by the applied hydrogen production technology (e.g. processing of hydrocarbons or water electrolysis) for those utilisation pathways relying on hydrogen as a precursor. As any other lifecycle activity, overall impacts will also be affected by the energy mix used to run CO<sub>2</sub> capture and subsequent utilisation processes to convert CO<sub>2</sub> into a useful product.

##### **4.4.5.1 Modelling requirements**

In line with the general provisions for system boundary setting (Section 3.2.3), the scope of any LCA of final CO<sub>2</sub>-based plastic products shall be "from cradle to grave". This means that, according to the modelling assumption prescribed below (i.e. raw gaseous CO<sub>2</sub> is a waste), activities occurring at the CO<sub>2</sub> source (power plant, cement production plant, etc.) after the CO<sub>2</sub> is generated shall be included in the system boundary (see also Section 4.2.1.5 for more detail). In addition, the End of Life of the CO<sub>2</sub>-based plastic product shall be considered, along with the intermediate stages of Manufacturing, Distribution and Use.

As a general rule, the CO<sub>2</sub> source shall be clearly specified. Similarly, the CO<sub>2</sub> conversion technology (i.e. the utilisation process) should be clearly described, specifying either its Technology Readiness Level (TRL) or at least its development level (research, experimental/laboratory scale, pilot scale, demonstration plant, etc.).

For modelling purposes, raw gaseous CO<sub>2</sub> arising from point emission sources shall be considered as a waste for recycling. This assumption is based on the discussion reported

in Annex G (Section G.1), and especially considering that raw CO<sub>2</sub> has no economic value before capturing (i.e. at the point of arising), and that for most existing CO<sub>2</sub> sources its availability is currently much higher compared to its demand for subsequent capture and utilisation (for polymer production and in general). However, such an assumption may need to be reconsidered in the future to reflect any change in availability of raw CO<sub>2</sub> from specific sources and its demand for utilisation purposes. Moreover, in some industrial processes (e.g. ammonia production) a concentrated CO<sub>2</sub> flow is regularly produced as an integral part of the process itself (e.g. in the case of ammonia production, separation of CO<sub>2</sub> contained in the gaseous mixture of CO<sub>2</sub> and H<sub>2</sub> generated in the process is regularly and necessarily carried out, providing an almost pure CO<sub>2</sub> stream). In this case, if the separated CO<sub>2</sub> has a positive market value (possibly after some additional compression/liquefaction), it shall be considered a co-product (and modelled according to the provisions reported in Section 4.5).

Leaving this exception apart, when raw CO<sub>2</sub> is considered a waste for recycling, the subsequent processes of capture, purification, possible compression/liquefaction and transport, as well as (depending on the pathway) also 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 (e.g. methanol or polyols), ultimately replacing an equivalent product from primary resources (e.g. conventional fossil-based sources). In other words, the CO<sub>2</sub>-based synthesis pathway (starting with capture or purification, depending on the CO<sub>2</sub> source<sup>80</sup>) can be assimilated to an "extended" recycling process (recycling chain), which turns waste CO<sub>2</sub> into a useful, more or less proximate CO<sub>2</sub>-based product.

As in any recycling situation, the Circular Footprint Formula (CFF; Section 4.4.10.2) shall be applied for modelling<sup>81</sup>. This means that all activities performed after the CO<sub>2</sub> is generated (capture/purification, possible transport, any additional processing, etc.) shall be modelled until a first useful CO<sub>2</sub>-based product is obtained which replaces an equivalent (conventional) product normally produced from primary (e.g. fossil-based) resources. Substitution at the point of substitution shall then be performed, and the CFF shall be applied (Figure 5).

The CFF shall be applied considering the most proximate and suitable point of substitution between a CO<sub>2</sub>-based material/product and an equivalent conventional primary material/product within the supply chain. Any relevant differences between the CO<sub>2</sub>-based synthesis pathway and the replaced conventional route shall be carefully evaluated and taken into account, as appropriate, in the modelling. This means that the point of substitution should be identified in correspondence of the first useful CO<sub>2</sub>-based product that is used as input to a same identical conversion/synthesis process in the supply chain as the replaced conventional product (as better discussed and exemplified in Annex G, Section G.2). Depending on the pathway, the CO<sub>2</sub>-based product considered for substitution can either be represented by captured CO<sub>2</sub> ready for use in a subsequent identical process, or by any downstream CO<sub>2</sub>-based product from further processing of captured CO<sub>2</sub> (possibly with other raw materials). For instance, CO<sub>2</sub>-based *polyols* may be assumed to replace fossil-based ones, while CO<sub>2</sub>-based *propylene* from the *methanol-to-olefin* route may be considered to replace conventional *propylene* from naphtha cracking. However, if the process directly using captured CO<sub>2</sub> as input is identical to the

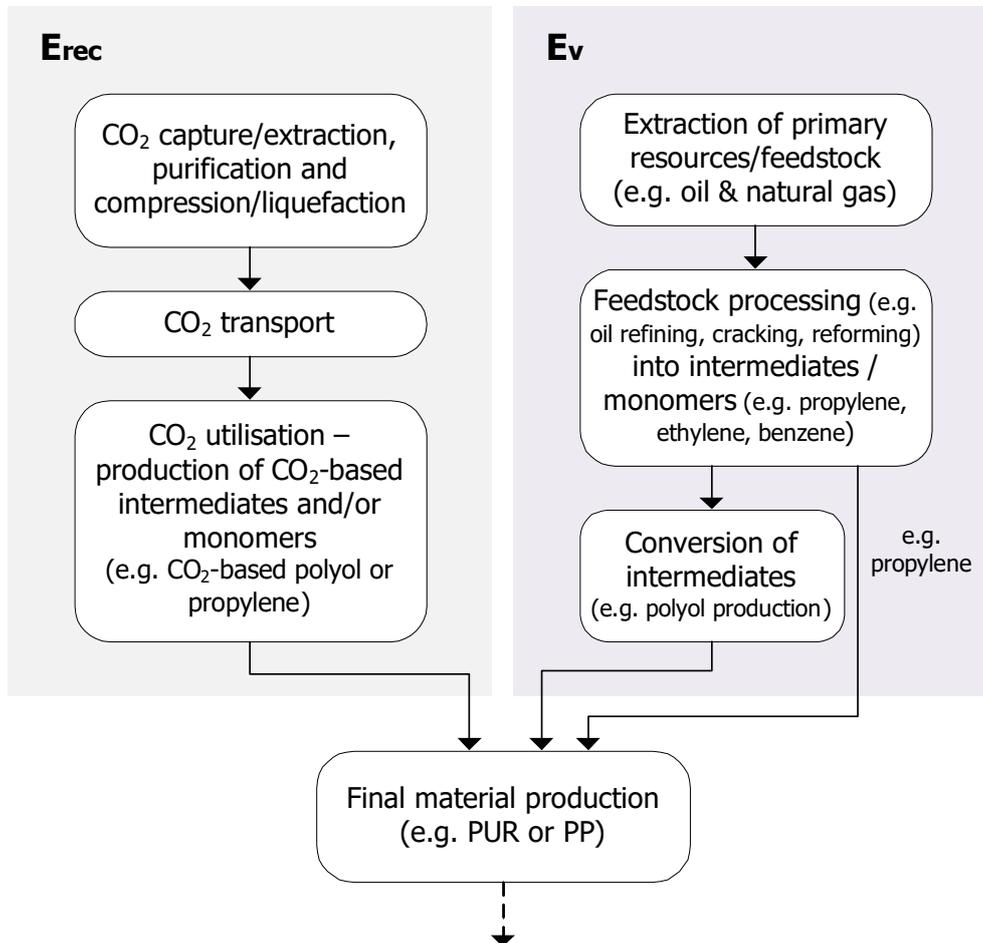
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<sup>80</sup> Capture shall be considered as the first step of the recycling chain in those cases where it is purposefully carried out to make the CO<sub>2</sub> available for downstream utilisation (e.g. in the case of power plants, cement kilns, incineration plants). When CO<sub>2</sub> separation (i.e. capture) from the gaseous stream it is part of is necessarily performed as an integral part of the process delivering the main product of the CO<sub>2</sub> source (e.g. during ammonia production, where CO<sub>2</sub> is traditionally separated from the mixture with H<sub>2</sub> to allow downstream use of the latter), the first step of the recycling chain shall coincide with purification of the concentrated CO<sub>2</sub> flow generated from the separation process. However, if the separated CO<sub>2</sub> already has a market value at that point (i.e. after separation and possible additional compression/liquefaction), it shall be treated as a co-product and modelled according to the provisions reported in Section 4.5.

<sup>81</sup> Note that the same approach (CFF) would apply even if raw CO<sub>2</sub> is considered a jointly produced co-product or by-product from the considered CO<sub>2</sub> source (as it is the case for most sources), and it has an economic value equal to zero or below at the point of arising.

process relying on the replaced conventional product, the point of substitution may be set at the level of captured CO<sub>2</sub> ready for use in such process.

No CO<sub>2</sub> uptake from air shall be modelled for captured CO<sub>2</sub> used in the investigated supply chain, if the considered CO<sub>2</sub> source relies on fossil-based feedstock.



**Figure 5.** Application of the Circular Footprint Formula (CFF) to model the use of CO<sub>2</sub> captured from gaseous effluents as a feedstock for plastic products and materials (e.g. CO<sub>2</sub>-based PUR or PP). The “A” factor shall be applied to model E<sub>rec</sub> and E<sub>v</sub>. In this example, R<sub>1</sub> is assumed equal to 1 (i.e. the product is entirely based on captured CO<sub>2</sub>).

The CFF applies the “A factor” to allocate the burdens from the recycling chain (i.e. the CO<sub>2</sub>-based pathway) and from the replaced primary production pathway between the system supplying and the one using the recycled material (i.e. captured CO<sub>2</sub> at the point of use or a downstream CO<sub>2</sub>-based product). The value of the “A factor” aims to reflect market realities, taking into account the relation between demand and supply of the recycled material. Since there is currently no established market for captured CO<sub>2</sub> ready for use and for derived CO<sub>2</sub>-based products (or no market is yet in place), the A factor shall be determined considering the specific market situation at the time of the study, applying the procedure described in Section 4.4.10.2.2 (in line with the PEF method). If the point of substitution is identified at the level of captured CO<sub>2</sub> ready for use in a subsequent process (i.e. when the downstream process chain is identical in both the CO<sub>2</sub>-based pathway and the replaced conventional route), the study shall take into account the relationship between the actual supply and demand of captured CO<sub>2</sub> (i.e. not of the CO<sub>2</sub> that could be potentially captured from existing sources). At present, a situation of equilibrium between supply and demand is likely to occur for captured CO<sub>2</sub> (i.e. any amount supplied would be entirely used to fulfil current needs of downstream users) and a value of the A factor equal to 0.5 should be assumed in that case. Wherever the point

of substitution is not at the level of captured CO<sub>2</sub>, but of any subsequent CO<sub>2</sub>-based product (e.g. polyol or propylene), the specific situation for that product shall be considered. For instance, a factor of 0.2 may be currently applied for the polyol substitution (supply of CO<sub>2</sub>-based polyols is lower than overall polyol demand).

#### **4.4.6 Handling of products based on maturing or emerging technologies**

A variety of feedstocks and materials can be currently used for the manufacturing of plastic products, and additional options may be available in the future. Processes and technologies used to supply or produce such feedstocks or materials may thus presents different levels of development and optimisation, and/or different scales of production. For example, products based on alternative feedstocks/materials generally relies on more recent processes/technologies having a lower level of optimisation and running at smaller scales compared to products based on more established fossil-based resources. These include, for instance, products based on more recent bio-based polymers such as Polybutylene Succinate (PBS), Polylactic Acid (PLA) and PLA-based polymers, as well as starch-based polymers (although commercial production of some of the latter has been in place for 20 years or more, e.g. PLA and starch-based polymers). Such maturing or evolving processes and technologies may thus undergo further improvement and/or upscaling (with related scale effects) in the future. In some cases, technologies or processes that are still at an early stage of development (emerging technologies) may be applied, and no products relying on such feedstocks or materials may be available on the market yet. This is, for instance, the case of Polyethylene Furanoate (PEF)-based products or CO<sub>2</sub>-based products (e.g. CO<sub>2</sub>-based olefins or CO<sub>2</sub>-based Polyurethane products).

Particular care shall be taken in performing any comparisons and/or comparative assertions among products or product scenarios including, within the respective supply chains, processes or technologies at different development/maturity levels and/or with different scales of production. Establishing whether comparisons and comparative assertions are feasible and can be properly conducted for a specific product category is one of the objectives of PEFCRs. Any new PEFCR developed based on the *Plastics LCA* method shall thus carefully take into account the existence of possible relevant differences in maturity of technologies and process scales in the supply chain of alternative products falling within the respective product category. Wherever comparisons and comparative assertions are allowed by a PEFCR, any non-previously identified relevant differences among compared products should be clearly acknowledged in a LCA study, and properly taken into account in the interpretation of the respective results, to avoid their misleading use and communication. In particular, the results of a comparative LCA study complying with a PEFCR based on this method should not be used to claim the ultimate environmental superiority of a product based on more established technologies/processes, with respect to equivalent products relying on less mature or evolving processes or technologies. At the same time, no optimised data calculated for theoretical processes should be applied throughout the upstream supply chain<sup>82</sup> of products based on maturing/evolving technologies, to claim their superiority compared to equivalent products based on more established technologies. For the product manufacturing process, this option is however prevented by the general requirement of using (measured) company-specific data from real facilities for modelling purposes (Section 4.6). For processes occurring further upstream in the supply chain (e.g. production of polymers or intermediates), such possibility is discouraged by assigning lower data quality ratings to data based on (qualified) estimates from theoretical calculations (see Section 4.7).

Products based on innovative or emerging technologies at an early stage of development (e.g. at pilot or pre-commercial demonstration level) are typically not available on the market, and can hence be assessed only within company-internal LCA studies, the results

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<sup>82</sup> Note that the product manufacturing process shall always be modelled based on (measured) company-specific data from real facilities (see Section 4.6).

of which will not be communicated externally. Such studies can be conducted only if (measured) company-specific data from real facilities are available for the product manufacturing process (according to the general requirements reported in Section 4.6). Moreover, involved technologies and processes should be at least at a Technology Readiness Level (TRL) of a pilot plant (i.e. from TRL 5 onwards). If comparisons are allowed by any existing relevant PEFCR for the specific product category, the same recommendations reported above for comparative studies among products relying on technologies at different maturity levels should be taken into account, when internally comparing the product in scope with another one based on more established technologies. In particular, such comparison should not be based on optimised theoretical process data for any upstream process within the product supply chain, to internally evaluate or demonstrate environmental superiority of the product in scope.

Additional general considerations on the assessment of products relying on maturing or emerging technologies, and related challenges, are available in Annex H.

#### 4.4.7 Transport and logistics

Important parameters that shall be taken into account when modelling transport include:

- **Transport type:** The type of transport, e.g. by land (truck, rail, pipe), by water (boat, ferry, barge), or air (airplane);
- **Vehicle type and fuel consumption:** The type of vehicle(s) used by each transport type, as well as the respective fuel consumption when fully loaded and empty. An adjustment shall be applied to the consumption of a fully-loaded vehicle according to its loading rate<sup>83</sup>;
- **Loading rate (=utilisation ratio):** Environmental impacts are directly linked to the actual loading rate, which shall therefore be considered;
- **Number of empty returns:** the number of empty returns (i.e. the ratio between the distance travelled to collect the next load after unloading the product and the distance travelled to transport the product), when applicable and relevant, shall be taken into account. The kilometres travelled by the empty vehicle shall be allocated to the product. In default transport datasets this is often already taken into account in the default utilisation ratio;
- **Transport distance:** Transport distances shall be documented, applying average transport distances specific to the context being considered;
- **Fuel production:** Fuel production shall be taken into account;
- **Infrastructure:** the transport infrastructure, that of road, rail and water shall be taken into account, unless they may be excluded based on Section 4.4.13 (on capital goods) or the cut-off criteria (Section 4.6.4);
- **Resources and tools:** the amount and type of additional resources and tools needed for logistic operations such as cranes and transporters shall be taken into account, unless they may be excluded based on the cut-off criteria (Section 4.6.4).

##### 4.4.7.1 Allocation of impacts from transport – truck transport

EF compliant LCI datasets for truck transport<sup>84</sup> are calculated for a reference flow of 1 tkm (tonne\*km) expressing the environmental burdens of 1 tonne of product that is transported for 1 km in a truck with a certain load. The transport payload (=maximum

<sup>83</sup> The loading rate is the ratio of actual load to the full load or capacity (e.g. mass or volume) that a vehicle carries per trip.

<sup>84</sup> Available at <http://lcdn.thinkstep.com/Node/>. EF compliant datasets are life cycle inventory datasets complying with the modelling rules specified in the PEF method, and with the respective requirements in terms of nomenclature, format and documentation to be used in a dataset. EF compliant datasets can be used for free to develop a LCA study under a specific PEFCR (Product Environmental Footprint Category Rule).

mass allowed) is indicated in the dataset. For example, the dataset related to transport with a truck of 28-32t considers the use of a truck with a full load mass of 28-32 t and a payload of 22t. The dataset for 1 tkm thus expresses the environmental burdens of 1 ton of product that is transported for 1 km within a 22t loaded (i.e. fully loaded) truck. The overall transport burdens (including emissions) for covering 1 km with a fully loaded truck are allocated to the reference flow of 1 tkm considered in the dataset based on the payload, so that only 1/22 share of the full burdens of the truck are accounted in the dataset for 1 tkm. When the mass of a full freight is lower than the maximum load capacity (payload) of the truck (e.g., 10t), the transport of the product may be considered volume limited. In this case, the truck has less fuel consumption and emissions per total load transported and the environmental burdens for the full load are 10/22 of the total burdens (emissions) of the volume limited truck. Therefore, the allocation of truck burdens shall be based on mass.

In EF compliant datasets<sup>85</sup> the transport payload is modelled in a parameterised way through the utilisation ratio. The utilisation ratio is calculated as the kg real load divided by the kg payload and shall be adjusted upon the use of the dataset. In case the real load is 0 kg, a real load of 1 kg shall be used to allow the calculation. Empty return trips may be included in the utilisation ratio by considering the % of empty km driven. For instance, if the truck is fully loaded for delivery but half empty at its return, the utilisation ratio is  $(22\text{t real load} / 22\text{t payload} * 50\% \text{ km} + 11\text{t real load} / 22\text{t payload} * 50\% \text{ km}) = 75\%$ .

*Plastics LCA* studies shall specify the utilisation ratio used for each truck transport modelled and clearly indicate whether the utilisation ratio includes empty return trips.

The following default utilisation ratios shall be used:

- If the load is mass-limited: a default utilisation ratio of 64%<sup>86</sup> shall be used, unless specific data is available. This default utilisation ratio includes empty return trips, which thus shall not be modelled separately.
- Bulk transport (e.g., gravel transport from mining pit to concrete plants) shall be modelled with a default utilisation ratio of 50% (100% loaded outbound and 0% loaded inbound), unless specific data is available.

#### **4.4.7.2 Allocation of impacts from transport – Van transport**

Vans are often used for home delivery of products like books and clothes or for home delivery from retailers. For vans, volume is the limiting factor rather than mass. Often, the van is half empty. If no specific information is available to perform the LCA study, a lorry with a full load mass of <1.2t with a default utilisation ratio of 50% shall be used. In case no dataset is available for a lorry of <1.2t, a lorry of <7.5t shall be used as approximation, with an utilisation ratio of 20%. A lorry of <7.5t with a payload of 3.3t and an utilisation ratio of 20% comes to the same load as a van of <1.2t and an utilisation ratio of 50%.

#### **4.4.7.3 Allocation of impacts from transport – Consumer transport**

Allocation of burdens from using passenger car for transport of goods shall be based on volume. The maximum volume to be considered for consumer transport is 0.2 m<sup>3</sup> (around 1/3 of a trunk of 0.6 m<sup>3</sup>). For products larger than 0.2 m<sup>3</sup> the full car transport burdens shall be considered. For products sold through supermarkets or shopping malls, the product volume (including packaging and empty spaces such as between fruits or bottles) shall be used to allocate the transport burdens between the products transported. The allocation factor shall be calculated as the volume of the product transported divided by 0.2 m<sup>3</sup>. To simplify the modelling, all other types of consumer

<sup>85</sup> Available at <http://lcdn.thinkstep.com/Node/>.

<sup>86</sup> Eurostat (2015) indicates that 21% of the km truck transport are driven with empty load and 79% are driven loaded (with an unknown load). In Germany only, the average truck load is 64%.

transport (like after buying in specialised shops or using combined trips) shall be modelled as if the product is purchased at a supermarket. The LCA study report shall specify the allocation value that has been used.

#### **4.4.7.4 Default scenarios – from supplier to factory**

If no specific data are available to perform the LCA study, then the default data provided below shall be used:

*For suppliers located within Europe:*

- For packaging materials from manufacturing plants to filler plants (beside glass; values based on Eurostat, 2015)<sup>87</sup>, the following scenario shall be used:
  - 230 km by truck (>32 t, EURO 4); and
  - 280 km by train (average freight train); and
  - 360 km by ship (barge).
- For transport of empty bottles, the following scenario shall be used:
  - 350 km by truck (>32 t, EURO 4); and
  - 39 km by train (average freight train); and
  - 87 km by ship (barge).
- For all other products from supplier to factory (values based on Eurostat, 2015)<sup>88</sup>, the following scenario shall be used:
  - 130 km by truck (>32 t, EURO 4); and
  - 240 km by train (average freight train); and
  - 270 km by ship (barge).

*For suppliers located outside of Europe:*

- 1000 km by truck (>32 t, EURO 4), for the sum of distances from harbour/ airport to factory outside and inside Europe; and
- 18,000 km by ship (transoceanic container) or 10,000 km by plane (cargo).

If producers' country (origin) is known: the adequate distance for ship and airplane should be determined using <https://www.searates.com/services/distances-time/> or [https://co2.myclimate.org/en/flight\\_calculators/new](https://co2.myclimate.org/en/flight_calculators/new).

In case it is not known whether the supplier is located within or outside of Europe, transport shall be modelled as if the supplier was located outside of Europe.

#### **4.4.7.5 Default scenarios – from factory to final client**

The transport from factory to final client (including consumer transport) shall be included in the Distribution stage of the LCA study. In case no specific information is available, the default scenario outlined below shall be used as a basis (see Figure 6). The following values shall be determined by the user of the *Plastics LCA* method (specific information shall be used, unless it is not available):

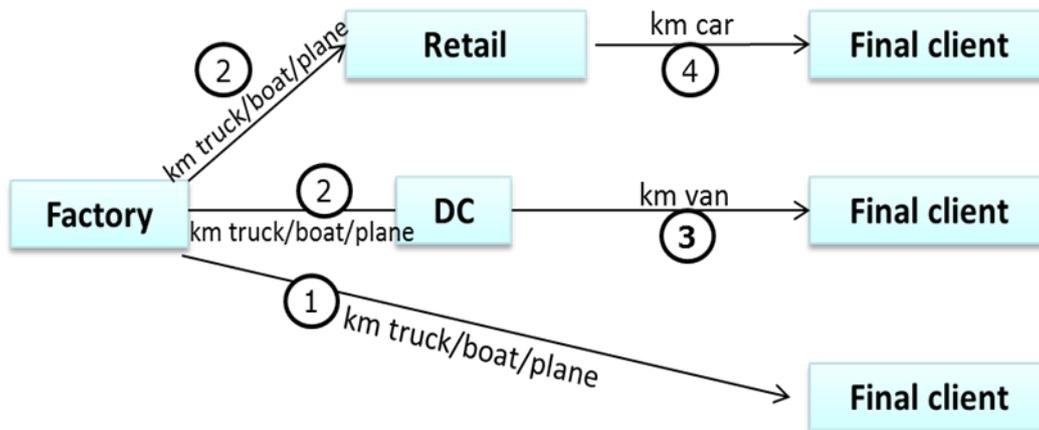
- Ratio between products sold through retail, distribution centre (DC) and directly to the final client;
- For factory to final client: Ratio between local, intracontinental and international supply chains;

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<sup>87</sup> Calculated as the mass weighted average of the goods categories 06, 08 and 10 using the Ramon goods classification for transport statistics after 2007. The category 'non-metallic mineral products' are excluded as they can double count with glass.

<sup>88</sup> Calculated as the mass weighted average of the goods of all categories.

- For factory to retail: distribution between intracontinental and international supply chains.



**Figure 6.** Default transport scenario.

The following combinations of transport modes, vehicle types and transport distances shall be used as default:

- X% from factory to final client:
  - X% local supply chain: 1,200 km by truck (>32 t, EURO 4);
  - X% intracontinental supply chain: 3,500 km by truck (>32 t, EURO 4);
  - X% international supply chain: 1,000 km by truck (>32 t, EURO 4) and 18'000 km by ship (transoceanic container). Note that for specific cases, plane or train may be used instead of ship.
- X% from factory to retail/ distribution centre (DC):
  - X% local supply chain: 1,200 km by truck (>32 t, EURO 4);
  - X% intracontinental supply chain: 3,500 km by truck (>32 t, EURO 4);
  - X% international supply chain: 1,000 km truck (>32 t, EURO 4), and 18'000 km by ship (transoceanic container). Note that for specific cases, plane or train may be used instead of ship.
- X% from DC to final client:
  - 100% Local: 250 km round trip by van (lorry <7.5t, EURO 3, utilisation ratio of 20%).
- X% from retail to final client:
  - 62%: 5 km, by passenger car (average);
  - 5%: 5 km round trip, by van (lorry <7.5t, EURO 3 with utilisation ratio of 20%);
  - 33%: no impact modelled

For reusable products the return transport from retail/DC to factory shall be modelled in addition to the transport needed to go to retail/DC. The same distances as for transport from product factory to final client shall be used (see above), however the truck utilisation ratio might be volume limited depending on the type of product.

Products frozen or cooled shall be modelled as transported in freezers or coolers.

#### 4.4.7.6 Default scenarios – from EoL waste collection to EoL treatment

The transport of product waste from the place of collection to End of Life treatment is normally included in the landfill and incineration EF compliant datasets provided by the EC<sup>89</sup>. It may also be included in available EF compliant recycling datasets. However, this shall be checked case by case, and additional default data may be needed in the LCA study. The following values shall be used in case no better data are available:

- Consumer transport from home to the place of collection: 1 km by passenger car;
- Transport from collection place to methanisation: 100 km by truck (>32 t, EURO 4);
- Transport from collection place to composting: 30 km by truck (lorry <7.5t, EURO 3).

#### 4.4.8 Storage at distribution centres or retail

Storage activities consume energy and potentially refrigerant gases. The following default data shall be used, unless more specific data are available:

Energy consumption at distribution centre: the storage energy consumption is 30 kWh/m<sup>2</sup>·year and 360 MJ bought (= burnt in boiler) natural gas (or 10 Nm<sup>3</sup> natural gas/m<sup>2</sup>·year plus related emissions from combustion). For centres that contain cooling systems, the additional energy use for the chilled or frozen storage is 40 kWh/m<sup>3</sup>·year (with an assumption of 2m height for the fridges and freezers). For centres with both ambient and cooled storage, 20% of the area of the DC is chilled or frozen. Note: the energy for chilled or frozen storage is only the energy to maintain the temperature.

Energy consumption at retail: a general energy consumption of 300 kWh/m<sup>2</sup>·year for the entire building surface shall be considered as default. For retail specialized in non-food/non-beverage products, a consumption of 150 kWh/m<sup>2</sup>·year for the entire building surface shall be considered. For retail specialized in food/beverage products, a consumption of 400 kWh/m<sup>2</sup>·year for the entire building surface plus energy consumption for chilled and frozen storage of 1,900 kWh/m<sup>2</sup>·year and 2700 kWh/m<sup>2</sup>·year respectively shall be considered (PERIFEM and ADEME, 2014).

Refrigerant gases consumption and leakages at DCs with cooling systems: gas content in fridges and freezers is 0.29 kg R404A per m<sup>2</sup> (Retail OEFSR<sup>90</sup>; Humbert et al., 2018). A 10% annual leakage is considered (Palandre, 2003). For the portion of refrigerant gases remaining in the equipment at end of life, 5% is emitted at end of life and the remaining fraction is treated as hazardous waste.

Only part of the emissions and resources emitted or used at storage systems shall be allocated to the product stored. This allocation shall be based on the space (in m<sup>3</sup>) and time (in weeks) occupied by the product stored. For this, the total storage capacity of the system shall be known, and the product-specific volume and storage time shall be used to calculate the allocation factor (as the ratio between product-specific volume\*time and storage capacity volume\*time).

An average DC is assumed to store 60,000 m<sup>3</sup> of product, out of which 48,000 m<sup>3</sup> for ambient storage and 12,000 m<sup>3</sup> for chilled or frozen storage. For a storage time of 52 weeks, a default total storage capacity of 3,120,000 m<sup>3</sup>\*weeks/year shall be assumed.

An average retail place is assumed to store 2000 m<sup>3</sup> of products (assuming 50% of the 2000 m<sup>2</sup> building is covered by shelves of 2 m height) during 52 weeks, i.e. 104,000 m<sup>3</sup> \* weeks/year.

<sup>89</sup> Available at <http://lcdn.thinkstep.com/Node/>.

<sup>90</sup> The OEFSR of the retail sector (v 1.0) is available at: [http://ec.europa.eu/environment/eussd/smgp/pdf/OEFSR-Retail\\_15052018.pdf](http://ec.europa.eu/environment/eussd/smgp/pdf/OEFSR-Retail_15052018.pdf).

#### **4.4.9 Use stage**

The Use stage is a life cycle stage that can provide a substantial contribution to the overall lifecycle impacts for many product categories. However, since the Use stage is often modelled based on several assumptions, the estimated contribution can be affected by even high uncertainty.

In cradle-to-grave LCA studies of final products, the Use stage shall always be included, following the guidelines outlined below. For final products, the LCIA results of the Use stage shall be reported separately and as sum with all other life cycle stages (total life cycle). The Use stage shall be excluded for cradle-to-gate LCA studies of intermediate products.

The Use stage of plastic products in many or most cases adds no major burdens to the overall lifecycle impacts, as it normally does not directly involve relevant energy consumption or direct emissions to the environment. However, in specific cases additives or other degradation compounds (including micro-plastics) may be released during the Use stage of plastic products (e.g. from outdoor furniture), potentially generating significant (toxicological) impacts even if emitted in small quantities. Such release shall thus be appropriately taken into account in the modelling (although no specific provisions can be given at this stage, and impacts from micro-plastic release are not directly covered in default impact categories applied in this method). Moreover, relevant material- or product-dependent aspects may substantially affect use-stage burdens (and impacts) of the product or service where a plastic product is used (e.g. a building, a vehicle, or delivered food in the case of plastic packaging). Affected use-stage burdens or activities in other product systems should hence be taken into account in the Use stage of a plastic product, especially when relevant differences may exist among product alternatives that may be compared under a specific PEFCR. For instance, the material used for an automotive panel may significantly affect the total fuel consumption of a vehicle, as it may happen for any generic plastic part used in mobile applications. If materials implying relevant differences in the ultimate mass of a panel or any other part of a vehicle are compared under a PEFCR, use-stage burdens related to fuel use in the vehicle (i.e. fuel production and emissions from use) should be taken into account, as well as the life cycle of connection parts to the rest of the vehicle, if subject to significant changes. Similarly, the type of insulation may affect consumption of construction materials in buildings (due to e.g. different wall thickness required to ensure the same thermal resistance), so that the life cycle of such materials should be modelled if used quantities significantly differ from one alternative to another. Also, the type of packaging may significantly affect refrigeration requirement for a given food product (due to e.g. different shelf lives) or the amount of energy needed to cook ready-to-prepare food (e.g. microwaving), which means that the burdens from refrigeration or cooking should be taken into account if there are significant differences among any alternatives compared under a specific PEFCR. Finally, the type of material used for an electronic device may affect its energy consumption, which should hence be modelled in case of important differences among alternatives.

##### **4.4.9.1 Types of use stage processes**

The Use stage often involves multiple processes. A distinction shall be made between (i) product independent and (ii) product dependent processes.

(i) **Product independent processes** have no relationship with the way the product in scope is designed or distributed. The Use stage process impacts will remain the same for all products in this product (sub)-category even if the producer changes the product's characteristics. Therefore, they do not contribute to any form of differentiation between two products or might even hide the difference. Examples are the use of a glass for drinking wine (considering that the product does not determine a difference in glass use); frying time when using olive oil; energy use for boiling one litre of water to be used for preparing coffee made from bulk instant coffee; the washing machine used for heavy laundry detergents (capital good).

(ii) **Product dependent processes** are directly or indirectly determined or influenced by the product design or are related to instructions for use of the product. These processes depend on the product characteristics and therefore contribute to differentiation between two products. All instructions provided by the producer and directed towards the consumer (through labels, websites or other media) shall be considered as product dependent. Examples of instruction are indications on how long the food must be cooked, how much water must be used, or in the case of drinks the recommended serving temperature and storage conditions. An example of a direct dependent process is the energy use of electric equipment when used in normal conditions.

Product dependent processes shall be included in the system boundary of the PEF study. Product independent processes shall be excluded from the system boundary and qualitative information may be provided.

#### **4.4.9.2 Main function approach or Delta approach**

Modelling of the Use stage may be done in different ways. Very often the related impacts and activities are modelled fully. For example, the total electricity consumption when using a coffee machine, or the total cooking time and related gas consumption when boiling pasta. In these cases, the Use stage processes for drinking coffee or eating pasta are related to the main function of the product (referred to as "main function approach").

In some cases, the use of one product can influence the environmental impact of another product. Some examples are:

- A toner cartridge is typically not held "responsible" for the paper it prints on. However, if a remanufactured toner cartridge works less efficiently and causes more paper loss compared to an original cartridge, the total (additional) cartridge-related paper loss should be considered. In that case, the paper loss is a product-dependent process of the Use stage of a remanufactured cartridge.
- The energy consumption during the Use stage of the battery/charger system is not related to the amount of energy stored and released from the battery. It only refers to the energy loss in each loading cycle. That energy loss can be caused by the loading system or the internal losses in the battery.

In these cases, only the additional activities and processes should be allocated to the product (e.g. paper and energy loss caused by remanufactured toner cartridge and battery, respectively). The method to deal with this multifunctionality consists in taking all associated products in the system (here paper and energy), and allocating the excess consumption of these associated products to the product that is considered responsible for this excess. This requires a reference consumption to be defined for each associated product (e.g. of energy and materials), which refers to the minimum consumption that is essential for providing the function. For instance, the reference consumption would be equal to 0 in the case of the remanufactured cartridge and related paper loss, and equally 0 for the battery. The consumption above this reference (the delta) will then be allocated to the product (Delta approach)<sup>91</sup>.

This approach should only be used for increasing impacts and to account for additional consumption above the reference.

To define the reference situation, the following sources shall be considered when available:

- Regulations applicable to the product (or product category) in scope;
- Standards or harmonised standards;
- Recommendations from manufacturers or manufacturers' organisations;
- Use agreements established by consensus in sector-specific working groups.

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<sup>91</sup> Specifications for drafting and revising product category rules (10.12.2014), ADEME.

The user of the *Plastics LCA* method shall decide the most appropriate approach to be taken (Main function approach or Delta approach), and shall describe it in the LCA report.

#### **4.4.9.3 Modelling the Use stage**

Annex D provides default data and assumptions that shall be used to model Use stage activities that might be crosscutting for several product categories. Better data should be used if available, and shall be made transparent and justified in the LCA report.

#### **4.4.10 End of Life modelling**

A general differentiation can be made between: (i) End of Life of the main product in scope once it has been discarded by the user entering the End of Life stage; and (ii) End of Life of the waste flows generated in the different upstream stages of the product life cycle before End of Life, i.e. during Raw Material Acquisition and Pre-processing, Manufacturing, Distribution, and Use of the product, as well as within all the related background activities (such as manufacturing of ingoing materials/parts, consumables and product packaging materials, energy supply, etc.).

End of Life of the waste flows generated during upstream life cycle stages occurring before End of Life (and in related background activities)) shall be modelled and reported at the specific stage where the waste occurs. For example, End of Life (treatment or disposal) of waste flows generated during manufacturing shall be modelled and reported at the Manufacturing stage. End of Life of any product losses occurring at the Distribution and Use stages shall also be included in the modelling and attributed to the specific stage where they occur (as better specified in Sections 4.2.3 and 4.2.4).

End of Life of the product in scope shall be modelled and reported at the End of Life stage. In general, the End of Life stage includes treatment or disposal of the product in scope after use, of the product possibly left at its end of use (such as food waste), and of any primary packaging of the product. For cradle-to-gate studies of intermediate products (e.g. polymers or unspecified plastic parts), the End of Life stage shall be excluded from the system boundary.

All waste flows arising throughout the product life cycle (including the product in scope at End of Life after use) shall be modelled up to the level of elementary flows. This means that waste flows shall not be included as such in the Life Cycle Inventory (e.g. as kg of household or hazardous waste generated), while the emissions to air, water and soil, and natural resource consumption resulting from their End of Life management (e.g. recycling, incineration or landfilling) shall be modelled in the inventory.

The End of Life stage and the end of life of waste flows generated during upstream life cycle stages shall be modelled using the Circular Footprint Formula (CFF), which is described in Section 4.4.10.2. The formula specifies how the burdens and possible benefits of common end of life options (including recycling/reuse, energy recovery, composting, anaerobic digestion and landfilling) shall be accounted for in the inventory. General requirements and recommendations on how to define appropriate End of Life scenarios to consistently apply the CFF for modelling are provided in Section 4.4.10.1.

Requirements and recommendations for the selection of suitable life cycle inventory datasets to be used for the modelling of specific end of life options applicable to plastic products, to develop (when needed) any new EF compliant material-specific dataset for such options, and on relevant product/waste-specific parameters to be considered for that purpose, are also provided in Sections 4.4.10.3–4.4.10.12. Addressed options include mechanical recycling, industrial composting, anaerobic digestion, incineration, landfilling, organic material use-on-land and in-situ biodegradation.

#### **4.4.10.1 End of Life scenarios**

A combination of different end of life options is generally applied to waste products and materials occurring throughout the product life cycle. This is especially the case of the product in scope, when it has reached its End of Life and is delivered to an advanced waste management system including separate collection of recoverable materials. An appropriate end of life scenario shall thus be defined for the product in scope, to model its End of Life stage via the Circular Footprint Formula (CFF). This scenario shall include viable end of life options actually applied to the product in scope within the reference geography and at the time of the study. The relative share of each option shall be based on most recent (year of analysis) practice, technology and data, taking consistently into account the specific rules to determine the "recycling output rate" ( $R_2$ ) and other relevant parameters used in the CFF, i.e. the incineration rate ( $R_3$ ), and the landfilling rate (calculated by difference as  $1-R_2-R_3$ ) (see Section 4.4.10.2.9).

When new statistics or values need to be generated or estimated<sup>92</sup>, these should be based on most recent European, national or regional waste statistics (depending on the geographical scope of the study), and/or data supplied by relevant industry associations or operators of extended producer responsibility schemes (e.g. packaging material consortia). Shares of single options should be determined as the average of shares individually calculated for each of the three most recent years for which data are available (or for any lower number of years for which they are provided).

For non-commercially available products, realistic end of life scenarios should be defined based on waste collection and treatment pathways implemented at the time of the study to the same product and/or material category, and which may be directly applied to the product in scope once it is introduced into the market. Relevant material properties shall be taken into account to identify viable options for the product in scope, such as its biodegradability under specific -controlled- conditions, which determines viability of composting and anaerobic digestion. Pathways that would be reasonably and easily implemented after commercialisation of the product may also be included in separate end of life scenarios, provided that suitable technology is already available to handle the relevant waste stream, or existing technologies/processes can be easily adapted to this purpose. For instance, mechanical recycling may be considered a viable option for a product (e.g. a bottle) based on a new polymer, if separate collection of products belonging to the same category is already in place (as it is the case of bottles), sorting technologies can be (easily) adapted to separate the new material stream without contaminating other streams already sorted out for recycling, and if existing (or similar) processes can be applied for recycling of the new material (e.g. no particular separation, purification/decontamination or compounding steps are needed) or a suitable recycling process is already available for that material. Shares of individual end of life options may be determined based on available data related to the same product category as the product in scope or, if possible and relevant, to the specific existing product(s) it intends to compete with. For instance, in the case of bottles manufactured from a new, non-biodegradable polymer, the same collection rate for recycling as bottles made of the relevant existing counterpart (e.g. PET or HDPE bottles) may be assumed, under the conditions reported above. Moreover, the relative incineration and landfilling shares of plastic packaging in general may be assumed (unless more specific data for bottles are available). If the new, non-commercially available product is only affected by a change in the feedstock used for an existing material (e.g. an alternative feedstock not yet produced commercially is used), the same end of life options and shares of the product manufactured from conventional feedstock should be assumed. All assumptions related to the applied end of life options and shares shall be clearly justified and documented in the LCA study report.

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<sup>92</sup> New statistics may be generated for  $R_2$  (recycling rate) when no company-specific values are available, nor default application-specific or material-specific values are reported in Annex C to the PEF method. For  $R_3$  (incineration rate) and the complementary landfilling rate ( $1-R_2-R_3$ ), new values need to be generally estimated, except when default values reported in Annex C for municipal waste are applicable.

At End of Life, a certain share of the product in scope may end up as littered into the environment as macro-plastic, either on-land or (directly or indirectly) into the riverine and/or marine environment. Littering does not represent an intended end of life option for (plastic) products, but rather a consequence of improper consumer behaviour and/or of waste mismanagement practices, which similarly to accidents are normally not considered in LCA. However, due to the current relevance of this environmental issue for plastic products, end of life scenarios should also account for the portion of product being littered, with the respective burdens and impacts (as far as suitable and agreed methods that are currently unavailable are developed to address these impacts). The estimated quantity of product ending up as macro-plastic released into the environment at End of Life (per functional unit) should also be reported as "additional environmental information" (as discussed in Section 3.2.5). Few data are currently available to estimate reliable and accurate product-specific littering rates and the resulting amount of product being released into the terrestrial, riverine and/or marine environment. However, for plastic products, preliminary estimates can be made based on most agreed and solid/complete approaches currently available, such as the one reported in the *Plastic Leak Project* (PLP) guidelines (Peano et al., 2020) to quantify leakage of plastics along the life cycle of products. This aspect is further addressed in Section 4.4.10.12 and Annex I.

#### 4.4.10.2 The Circular Footprint Formula

This section describes the Circular Footprint Formula (CFF), the parameters to be used, and how the formula and its parameters shall be applied to final products and to intermediate products. A fictional example of application is also reported at the end of this section.

The CFF is a combination of "material recovery + energy recovery+ disposal", i.e.:

##### Material recovery:

$$(1 - R_1)E_V + R_1 \times \left( A E_{recycled} + (1 - A)E_V \times \frac{Q_{Sin}}{Q_P} \right) + (1 - A)R_2 \times \left( E_{recyclingEoL} - E_V^* \times \frac{Q_{Sout}}{Q_P} \right)$$

##### Energy recovery:

$$(1 - B)R_3 \times (E_{ER} - LHV \times X_{ER,heat} \times E_{SE,heat} - LHV \times X_{ER,elec} \times E_{SE,elec})$$

##### Disposal:

$$(1 - R_2 - R_3) \times E_D \quad \text{[Equation 2]}$$

#### 4.4.10.2.1 Parameters of the CFF

**A:** allocation factor of burdens and credits between supplier and user of recycled materials.

**B:** allocation factor of energy recovery processes. It applies both to burdens and credits.

**Q<sub>Sin</sub>:** quality of the ingoing secondary material, i.e. the quality of the recycled material at the point of substitution.

**Q<sub>Sout</sub>:** quality of the outgoing secondary material, i.e. the quality of the recyclable material at the point of substitution.

**Q<sub>p</sub>:** quality of the primary material, i.e. quality of the virgin material.

**R<sub>1</sub>:** it is the proportion of material in the input to the production that has been recycled from a previous system.

**R<sub>2</sub>**: it is the proportion of the material in the product that will be recycled (or reused) in a subsequent system. R<sub>2</sub> shall therefore take into account the inefficiencies in the collection and recycling (or reuse) processes. R<sub>2</sub> shall be measured at the output of the recycling plant.

**R<sub>3</sub>**: it is the proportion of the material in the product that is used for energy recovery at EoL.

**E<sub>recycled</sub> (E<sub>rec</sub>)**: specific emissions and resources consumed (per functional unit) arising from the recycling process of the recycled (reused) material, including collection, sorting and transportation process.

**E<sub>recyclingEoL</sub> (E<sub>recEoL</sub>)**: specific emissions and resources consumed (per functional unit) arising from the recycling process at EoL, including collection, sorting and transportation process.

**E<sub>v</sub>**: specific emissions and resources consumed (per functional unit) arising from the acquisition and pre-processing of virgin material<sup>93</sup>.

**E\*v**: specific emissions and resources consumed (per functional unit) arising from the acquisition and pre-processing of virgin material assumed to be substituted by recyclable materials.

**E<sub>ER</sub>**: specific emissions and resources consumed (per functional unit) arising from the energy recovery process (e.g. incineration with energy recovery, landfill with energy recovery, etc.).

**E<sub>SE,heat</sub> and E<sub>SE,elec</sub>**: specific emissions and resources consumed (per functional unit) that would have arisen from the specific substituted energy source, heat and electricity respectively.

**E<sub>D</sub>**: specific emissions and resources consumed (per functional unit) arising from disposal of waste material at the EoL of the analysed product, without energy recovery.

**X<sub>ER,heat</sub> and X<sub>ER,elec</sub>**: the efficiency of the energy recovery process for both heat and electricity.

**LHV**: lower heating value of the material in the product that is used for energy recovery.

Users of the *Plastics LCA* method shall report all the parameters used. Default values for some parameters (A, R<sub>1</sub>, R<sub>2</sub>, R<sub>3</sub>, and Q<sub>s</sub>/Q<sub>p</sub> for packaging) are available in Annex C<sup>94</sup> to the PEF method, which is consistently implemented also in this document. Users of the *Plastics LCA* method shall report the version of Annex C they are referring to in the LCA study. Annex C is available at <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>.

If a default value for R<sub>1</sub> and R<sub>2</sub> is not included in Annex C, users of the *Plastics LCA* method may provide new values to the Commission. Such values shall be part of a study that has been reviewed by an external independent third party reviewer. The Commission will take the decision if the new values are acceptable and can be implemented in an updated version of Annex C.

#### **4.4.10.2.2 The A factor**

The A factor allocates burdens and credits from recycling and virgin material production between two life cycles (i.e. the one supplying and the one using recycled material) and it aims to reflect market realities.

<sup>93</sup> In case of open-loop recycling situations, the parameter E<sub>v</sub> reported in the term “(1-A)E<sub>v</sub> × Q<sub>s</sub><sub>in</sub>/Q<sub>p</sub>” of the CFF represents the burdens from acquisition and pre-processing of the virgin material assumed to be substituted by the recycled material (e.g. of wooden or concrete parts of street benches potentially replaced by profiled bars made of mixed recycled polyolefin waste).

<sup>94</sup> The list of values in Annex C is periodically reviewed and updated by the European Commission; users of the *Plastics LCA* method are invited to check and use the most updated values provided in the Annex.

An A factor equal to 1 would reflect a 100:0 approach (i.e. credits are given to the recycled content), an A factor equal to 0 would reflect a 0:100 approach (i.e. credits are given to the recyclable materials at the end of life).

In *Plastics LCA* studies the A factor values shall be in the range  $0.2 \leq A \leq 0.8$ , to always capture both aspects of recycling (recycled content and recyclability at end of life).

The driver to determine the values of the A factor is the analysis of the market situation. This means:

- **A=0.2.** Low offer of recycled materials and high demand: the formula focuses on recyclability at end of life.
- **A=0.8.** High offer of recycled materials and low demand: the formula focuses on recycled content.
- **A=0.5.** Equilibrium between offer and demand: the formula focuses both on recyclability at end of life and recycled content.

Default application-specific and material-specific A values are available in Annex C. The following procedure shall be applied (in hierarchical order) to select the value of A to be used in a *Plastics LCA* study:

- Check in Annex C the availability of an application-specific A value which fits the LCA study,
- If an application-specific A value is not available, the material-specific A value in Annex C shall be used,
- If a material-specific A value is not available, the A value shall be set equal to 0.5.

#### **4.4.10.2.3 The B factor**

The B factor is used as an allocation factor of energy recovery processes. It applies both to burdens and credits. Credits refer to the amount of heat and electricity sold, not to the total produced amount, taking into account relevant variations over a 12-months period, e.g. for heat.

In *Plastics LCA studies* the B value shall be set equal to 0 as default.

To avoid double-counting between the current and the subsequent system in case of energy recovery (i.e. between the system generating energy from waste and the one using recovered energy), the subsequent system shall model its own use of any recovered energy as primary energy.

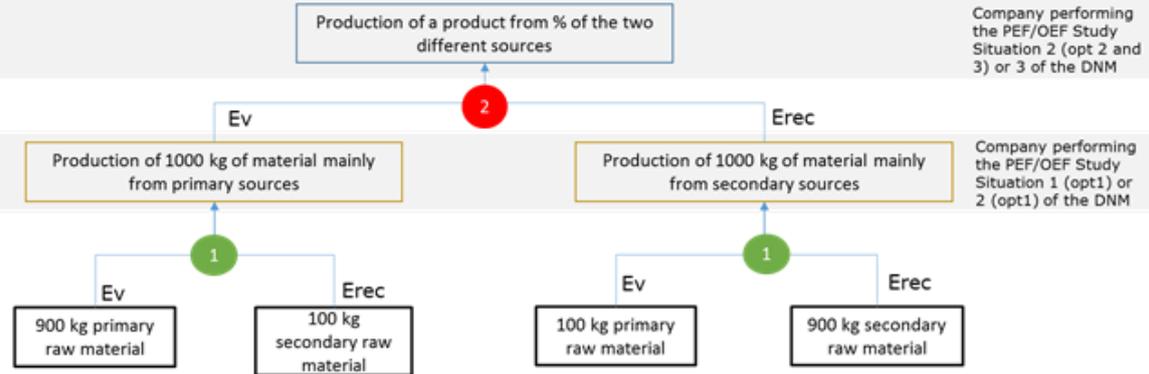
#### **4.4.10.2.4 The point of substitution**

It is necessary to determine the point of substitution to apply the “material” part of the formula. The point of substitution corresponds to the point in the supply chain where secondary materials substitute primary materials.

The point of substitution shall be identified in correspondence to the process where input flows are coming from 100% primary sources and 100% secondary sources (level 1 in Figure 7). This corresponds to, e.g. metal scrap, glass cullet and pulp input to the recycling process. However, in some cases, the point of substitution may be identified after some mixing of primary and secondary material flows has occurred (level 2 in Figure 7), which corresponds to e.g. metal ingots, glass and paper. In fact for glass the most appropriate point of substitution will be at level 2, as the glass making process typically differs when secondary or primary material is used (glass cullet is not equivalent to primary glass ingredients, but better).

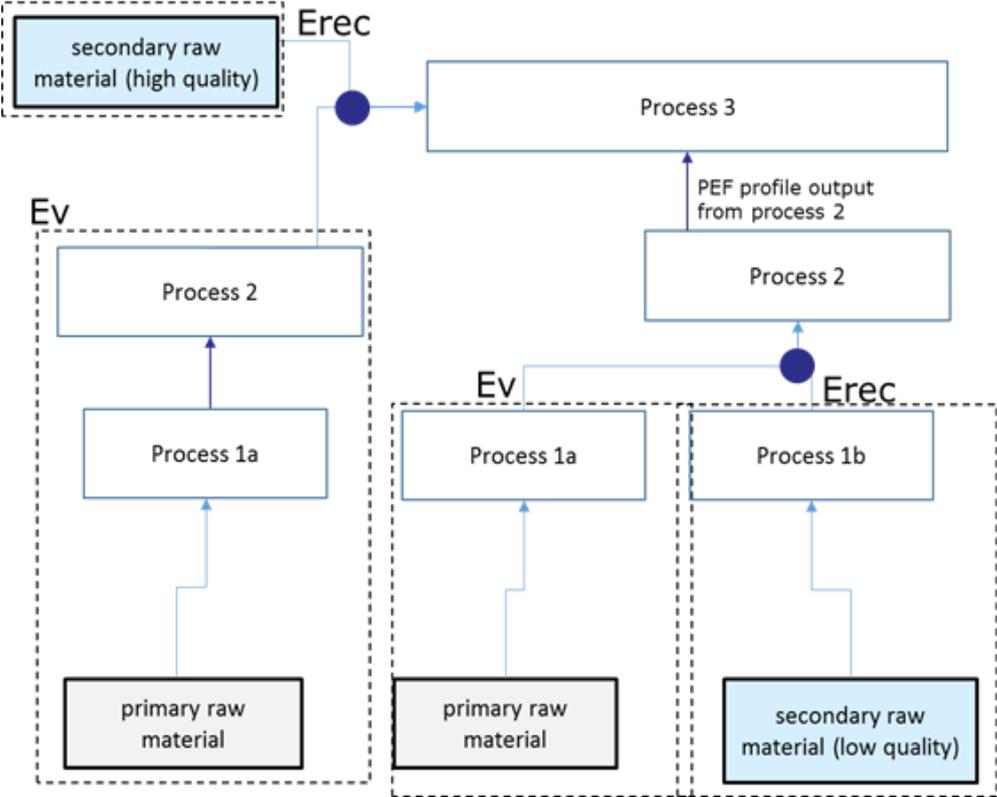
The point of substitution at level 2 may be applied only if the datasets used to model e.g.  $E_{rec}$  and  $E_v$  take into account the real (average) flows regarding primary and secondary materials. For example, if  $E_{rec}$  corresponds to the “production of 1 t of secondary material” (see Figure 7) and it has an average input of 10% from primary raw materials,

the amount of primary materials, together with their environmental burdens, shall be included in the  $E_{rec}$  dataset.



**Figure 7.** Point of substitution at level 1 and at level 2.

Figure 7 is a schematic representation of a generic situation (flows are 100% primary and 100% secondary). However, in practice in some situations more than one point of substitution can be identified at different steps in the value chain, as represented in Figure 8, where e.g. scrap of two different qualities is processed at different steps.



**Figure 8.** Example of point of substitutions at different steps in the value chain.

**4.4.10.2.5 The quality ratios:  $Q_{sin}/Q_p$  and  $Q_{sout}/Q_p$**

Two quality ratios are used in the CFF, to take into account the quality of both the ingoing and the outgoing recycled materials.

Two further cases are distinguished:

- (a) **If  $E_v = E^*_v$** , the two quality ratios are needed:  $Q_{S_{in}}/Q_p$  associated to the recycled content, and  $Q_{S_{out}}/Q_p$  associated to recyclability at EoL. The quality factors are there to capture the downcycling of a material compared to the original primary material and, in some cases, may capture the effect of multiple recycling loops.
- (b) **If  $E_v \neq E^*_v$** , one quality ratio is needed:  $Q_{S_{in}}/Q_p$  associated to the recycled content. In this case  $E^*_v$  refers to the functional unit of the material substituted in a specific application. For example, plastic recycled to produce a bench modelled via substitution of cement shall take into account also the "how much", "how long" and "how well". Therefore, the  $E^*_v$  parameter indirectly integrates the  $Q_{S_{out}}/Q_p$  parameter, and therefore the  $Q_{S_{out}}$  and  $Q_p$  parameters are not part of the CFF.

The quality ratios shall be determined at the point of substitution and per application or material.

The quantification of the quality ratios shall be based on:

- Economic aspects: i.e. price ratio of secondary compared to primary materials at the point of substitution. In case the price of secondary materials is higher than that of the primary ones, the quality ratios shall be set equal to 1.
- When economic aspects are less relevant than physical aspects, the latter may be used.

Packaging materials used by industry are often the same within different sectors and product groups: Annex C provides one worksheet with  $Q_{S_{in}}/Q_p$  and  $Q_{S_{out}}/Q_p$  values applicable to packaging materials. The company performing a *Plastics LCA* study may use different values and they shall be made transparent and justified in the LCA report.

#### **4.4.10.2.6 Recycled content ( $R_1$ )**

The  $R_1$  values applied shall be supply-chain or application-specific, depending on the information accessible by the company performing the LCA study. Default application specific  $R_1$  values are available in Annex C. The following procedure shall be applied (in hierarchical order) to select the value of  $R_1$  to be used in a LCA study:

- Supply-chain specific values shall be used when the process is run by the company performing the LCA study or when the process is not run by the company performing the LCA study but the company has access to (company)-specific information. (Situation 1 and Situation 2 of the Data Needs Matrix, see Section 4.7.5).
- In all other cases, the default secondary  $R_1$  values of Annex C (application-specific) shall be applied.  $R_1$  shall be set to 0% when no application-specific value is available.
- Material-specific values based on supply market statistics are not accepted as a proxy and therefore shall not be used.

The applied  $R_1$  values shall be subject to verification of the LCA study.

#### **4.4.10.2.7 Guidelines when using supply chain specific $R_1$ values**

When using supply-chain specific  $R_1$  values other than 0, traceability throughout the supply chain is mandatory. The following general guidelines shall be followed:

- The supplier information (through e.g., statement of conformity or delivery note) shall be maintained during all stages of production and delivery at the converter.
- Once the material is delivered to the converter for production of the end products, the converter shall handle information through their regular administrative procedures.
- The converter for production of the end products claiming recycled content shall demonstrate through its management system the [%] of recycled input material into the respective end product(s).

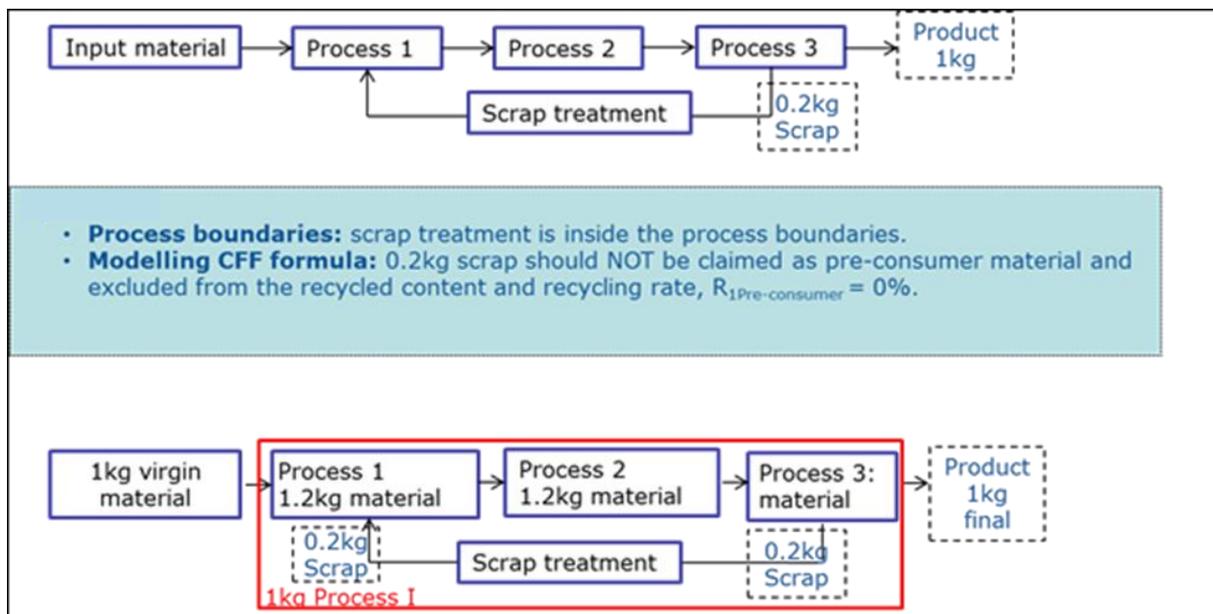
- The latter demonstration shall be transferred upon request to the user of the end product. In case a LCA profile is calculated and reported, this shall be stated as additional technical information of the LCA study.
- Industry- or company-owned traceability systems may be applied as long as they cover the general guidelines outlined above. If not, they shall be supplemented with the general guidelines above.

For the packaging industry, the following industry-specific guidelines are recommended:

- For the container glass industry (FEVE - The European Container Glass Federation): the European Commission regulation no 1179/2012<sup>95</sup>. This regulation requests a statement of conformity delivered by the cullet producer.
- For the paper industry: European Recovered Paper Identification System (CEPI – Confederation of European Paper Industries, 2008). This document prescribes rules and guidance on necessary information and steps, with a delivery note that shall be received at the reception of the mill.
- For beverage cartons no recycled content is used so far and thus sector specific rules are redundant for the moment. However, if needed, the same guidelines as for paper shall be used as being most suitable (beverage cartons are covered by a recovered paper grade category under EN643).
- For the plastics industry: EN standard 15343:2007. This standard prescribes rules and guidelines on traceability. The supplier of the recyclate is requested to provide specific information.

#### 4.4.10.2.8 Guidelines on how to deal with pre-consumer scrap

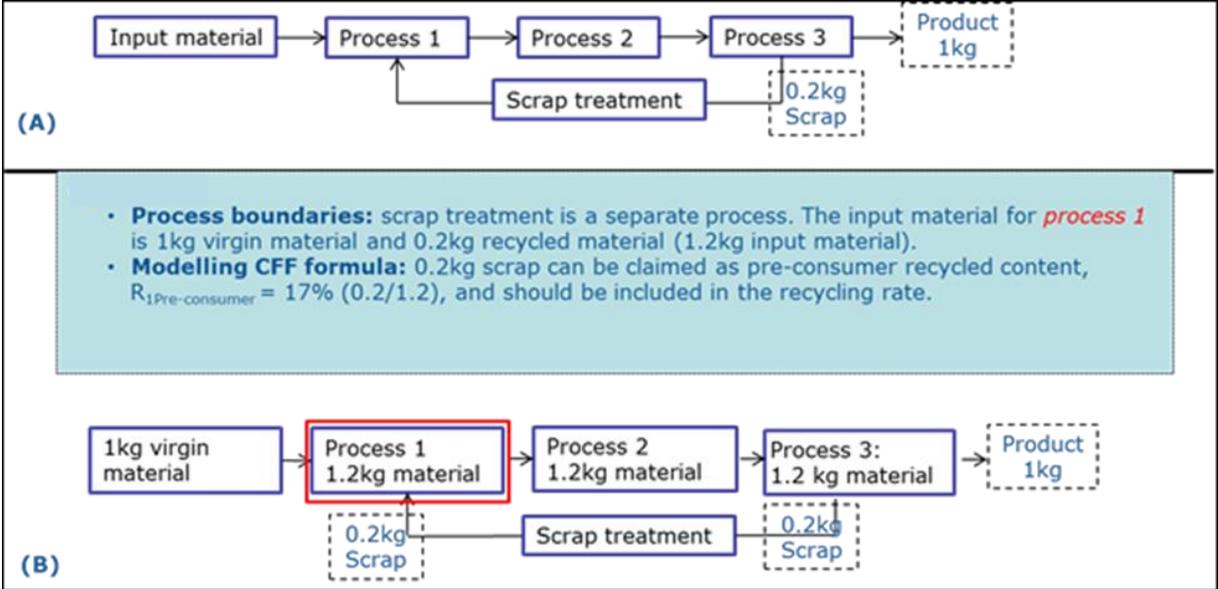
Any scrap material that circulates within a process chain or pool of process chains (pre-consumer scrap) should be excluded from being defined as recycled content and should not be included in  $R_1$ . Scrap should hence not be claimed as pre-consumer recycled content. Process boundaries and modelling requirements applying the Circular Footprint Formula are shown in Figure 9.



**Figure 9.** Modelling of pre-consumer scrap when it is not claimed as pre-consumer recycled content.

<sup>95</sup> Commission Regulation (EU) No 1179/2012 of 10 December 2012 establishing criteria determining when glass cullet ceases to be waste under Directive 2008/98/EC of the European Parliament and of the Council.

In specific situations, to be evaluated on a case-by-case basis and submitted to the LCA study verification, scrap may be claimed as pre-consumer recycled content and included in  $R_1$ . In this case, the impacts to produce the input material that lead to the pre-consumer scrap in question have to be allocated to the product system that generated this scrap. Process boundaries and modelling requirements applying the Circular Footprint Formula are shown in Figure 10.

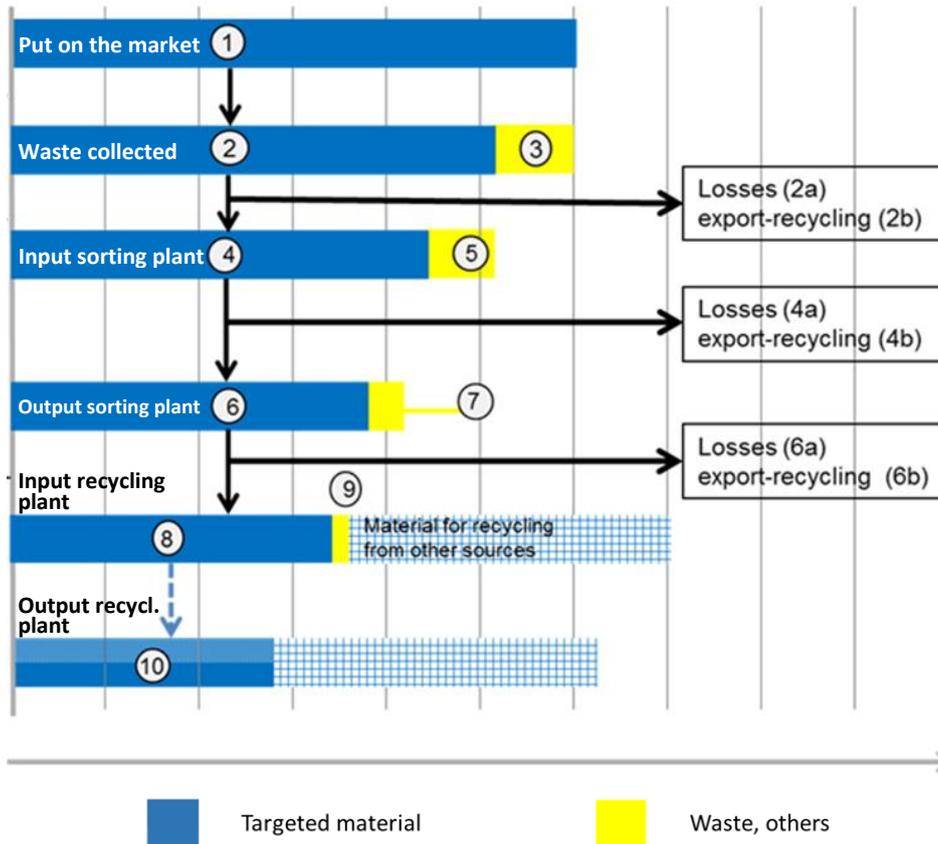


**Figure 10.** Modelling of pre-consumer scrap when it is claimed as pre-consumer recycled content.

**4.4.10.2.9 Recycling output rate ( $R_2$ )**

The  $R_2$  parameter refers to the “recycling output rate”: Figure 11 provides a visual representation. Often, values are available for point 8<sup>96</sup> in Figure 11 (i.e. at the input of recycling plants) or even for point 2 (i.e. for waste material collected for recycling). Therefore, such values shall be corrected to the actual output recycling rate (point 10) taking into account possible process losses sorting/separation and/or recycling. In Figure 11 the output recycling rate ( $R_2$ ) is in correspondence of point 10.

<sup>96</sup> Statistical data gathered in correspondence of point 8 in Figure 11 may be used to inform the calculation of the recycling output rate. Point 8 corresponds to recycling targets calculated according to the general rule provided in Directive (EU) 2018/851 of 30 May 2018. In some cases, under strict conditions and by way of derogation from the general rule, data may be available at point 6 in Figure 11 and may be used to inform the calculation of the recycling output rate.



**Figure 11.** Simplified collection and recycling scheme of a targeted waste material.

The product design and composition will determine if the material in the specific product is actually suitable for recycling. Therefore, before selecting the appropriate  $R_2$  value, an evaluation of the recyclability of the material shall be made and the LCA study shall include a statement on the recyclability of the materials/ products:

The statement on recyclability shall be provided together with an evaluation for recyclability that includes evidence for the following three criteria (as described by ISO 14021:2016, section 7.7.4 'Evaluation methodology'):

1. The collection, sorting and delivery systems to transfer the materials from the source to the recycling facility are conveniently available to a reasonable proportion of the purchasers, potential purchasers and users of the product;
2. The recycling facilities are available to accommodate the collected materials;
3. Evidence is available that the product for which recyclability is claimed is being collected and recycled.

If one criterion is not fulfilled, or the sector-specific recyclability guidelines indicate limited recyclability, an  $R_2$  value of 0% shall be applied. Point 1 and 3 may be proven by recycling statistics (country specific) derived from industry associations or national bodies. Approximation to evidence at point 3 may be provided by applying for example the design for recyclability evaluation outlined in EN 13430:2004 Material recycling (Annexes A and B) or other sector-specific recyclability guidelines if available. In this case, for PET bottles the EPBP (European PET Bottle Platform) design guidelines should be used (<https://www.epbp.org/design-guidelines>), while national guidelines or recommendations should be used for other plastic products (e.g. Recoup for UK, Cotrep and Cerec for France, Interseroh, Cirpack and Cyclos for Germany, etc.).

Default application-specific  $R_2$  values are available in Annex C. The following procedure shall be followed to select the  $R_2$  value to be used in a LCA study:

- Company-specific values shall be used when available and following the evaluation of recyclability.
- If no company-specific values are available and the criteria for the evaluation of recyclability are fulfilled (see above), application-specific  $R_2$  values shall be used selecting the appropriate value available in Annex C:
  - If an  $R_2$  value is not available for a specific country, then the European average shall be used;
  - If an  $R_2$  value is not available for a specific application, the  $R_2$  values of the material shall be used (e.g. materials' average);
  - In case no  $R_2$  values are available,  $R_2$  shall be set equal to 0 or new statistics may be generated in order to assign an  $R_2$  value in the specific situation.

The applied  $R_2$  values shall be subject to the LCA study verification.

Background information to calculate the  $R_2$  values for packaging materials is available in Annex C.

#### **4.4.10.2.10 $E_{recycled}$ ( $E_{rec}$ ) and $E_{recyclingEoL}$ ( $E_{recEoL}$ )**

The system boundary of  $E_{rec}$  and  $E_{recEoL}$  shall consider all the emissions and resources consumed starting from collection of the waste material up to the defined point of substitution.

If the point of substitution is identified at "level 2"  $E_{rec}$  and  $E_{recEoL}$  shall be modelled using the real input flows. Therefore, if a portion of the input flows are from primary raw materials, they shall be included in the datasets used to model  $E_{rec}$  and  $E_{recEoL}$ .

In some cases  $E_{rec}$  may correspond to  $E_{recEoL}$ , for example in cases where closed loops occurs.

#### **4.4.10.2.11 The $E^*_v$**

When default  $E^*_v$  equals  $E_v$ , the user shall assume that a recyclable material at End of Life is replacing the same virgin material which was used at the input side to produce the recyclable material.

In some cases  $E^*_v$  will be different from  $E_v$ . In this case, the user shall provide evidence that a recyclable material is substituting a different virgin material than the one producing the recyclable material.

If  $E^*_v \neq E_v$ ,  $E^*_v$  represents the actual amount of virgin material substituted by the recyclable material. In such cases  $E^*_v$  is not multiplied by  $Q_{Sout}/Q_p$ , because this parameter is indirectly taken into account when calculating the "actual amount" of virgin material substituted. Such amount shall be calculated taking into account that the virgin material substituted and the recyclable material fulfil the same function in terms of "how long" and "how well".  $E^*_v$  shall be determined based on evidence of actual substitution of the selected virgin material.

#### **4.4.10.2.12 How to apply the formula to intermediate products (cradle-to-gate studies)**

In cradle-to-gate LCA studies of intermediate products, the End of Life of the product (i.e. recyclability at end-of-life, energy recovery, disposal) shall not be accounted for, unless an applicable (future) PEFCRs would require to calculate additional environmental information for the End of Life stage.

If the CFFs applied in LCA studies of intermediate products incorporating recycled content (cradle-to-gate studies), the user shall:

- Use Equation 3;
- Exclude the End of Life by setting the parameters  $R_2$ ,  $R_3$ , and  $E_d$  to 0 for the product(s) in scope;
- Use and report the results with two A values for the product in scope i.e. :
  - Setting  $A = 1$ : to be used as default in the LCA profile calculation. The purpose of this setting is to allow to focus the hotspot analysis on the actual system,
  - Setting  $A =$  the application- or material-specific default values as listed in Annex C.

The results calculated considering default application- or material-specific values shall be reported as 'additional technical information' and used when creating EF compliant datasets. This allows using a correct A value when the dataset is used in future modelling.

#### **4.4.10.2.13 How to deal with specific aspects**

##### **Recovery of bottom ashes or slag from incineration**

Recovery of bottom ashes/slag shall be included in the  $R_2$  value (recycling output rate) of the original product/material. Emissions from their treatment is within the  $E_{recEoL}$ .

##### **Landfill and incineration with energy recovery**

Whenever a process, such as landfill with energy recovery or municipal solid waste incineration with energy recovery, is leading to an energy recovery, it shall be modelled under the "energy" part of the CFF. The credit is calculated based on the amount of output energy that is used outside the process.

##### **Municipal solid waste**

Annex C provides default values per country (i.e. relative incineration and landfilling shares within single EU countries) that shall be used to quantify the share of product waste sent to incineration ( $R_3$ ) and the share sent to landfill ( $1-R_2-R_3$ ), unless supply-chain specific values are available.

##### **Compost and anaerobic digestion/sewage treatment**

Compost, including digestate coming out of the anaerobic digestion, shall be treated in the "material" part of the CFF like a case of recycling with  $A = 0.5$ , and with  $E^*_v$  representing the (virgin) production burdens of the product(s) replaced from compost (if any, e.g. mineral/synthetic fertilisers and/or peat). If composting or digestion is applied to specific biodegradable (plastic) products, substitution of mineral fertilisers shall be assumed only when the product in scope actually contains relevant nutrients (e.g. N, P and/or K) in its composition, and these are ultimately transferred into the residual organic material obtained from treatment. The actual amount of product replaced shall be considered for the calculation of  $E^*_v$  (based on e.g. the content of available nutrients in the residual organic material, compared to replaced mineral fertilisers).

The energy part of anaerobic digestion shall be treated as a normal process of energy recovery under the "energy" part of the CFF.

##### **Waste materials used as a fuel**

When a waste material is used as a fuel (e.g. waste plastic used as fuel in cement kilns), it shall be treated as an energy recovery process under the "energy" part of the CFF.

### **Modelling complex products**

When considering complex products (e.g. printed wiring boards) with complex End of Life management, the default values of the parameters shall refer to the ones in Annex C and shall be available as metadata information in the dataset if it is foreseen to have an aggregated dataset. The Bill of Material (BoM) should be taken as a starting point for calculations if no default data is available.

### **Reuse and refurbishment**

If the reuse/refurbishment of a product results in a product with different product specifications (providing another function than the product being reused/refurbished), this shall be considered as part of the CFF, as a form of recycling (see Section 4.4.11). Also, old parts that have been changed during refurbishment shall be modelled under the CFF.

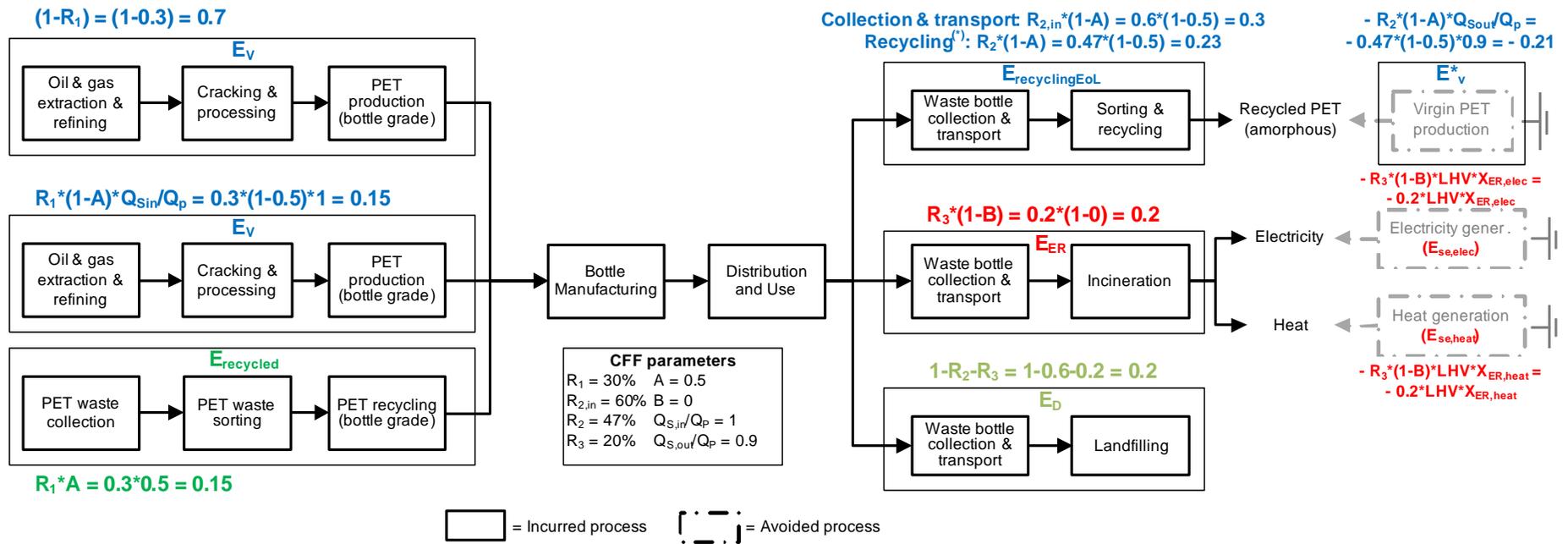
In this case, reuse/refurbishment activities are part of the  $E_{recEoL}$  parameter, while the alternative function provided (or the avoided production of parts or components) falls under the  $E^*v$  parameter.

#### **4.4.10.2.14 Example**

Figure 12 provides a fictional example of application of the CFF to the life cycle of a disposable PET beverage bottle. This is assumed to incorporate 30% recycled content ( $R_1 = 30\%$ ; supply-chain specific value appropriately evidenced by the company conducting the LCA study), and to be collected for recycling at End of Life at a rate of 60% (based on ICIS and Petcore Europe, 2018). For clarity, this collection rate is indicated as  $R_{2,in}$  in Figure 12, as useful to model PET waste collection and transport processes (whose burdens are generally provided per unit mass of waste to be collected/transported). The actual output recycling rate ( $R_2$ ) applied in the CFF is calculated assuming a sorting efficiency of 91%, and a recycling efficiency equal to 85.5%, leading to an ultimate recycling rate at the output of the recycling plant equal to 47% (i.e.  $R_2 = 47\%$ ; supply-chain specific value supported by specific evidence in the LCA study). The incineration rate ( $R_3$ ) is assumed at 20%, with the resulting landfilling rate ( $1-R_2-R_3$ ) also equalling 20% (based on data related to plastic packaging waste management in the EU over the years 2014-2016; Eurostat, 2019). The A parameter is set to 0.5, according to the default application- and material-specific values reported in Annex C for PET bottles and generic PET resin, while the B parameter is set to 0 as default. Default quality ratios from Annex C are also applied for both the recycled content ( $Q_{Sin}/Q_p$ ; equal to 1 for bottle grade PET from solid state poly-condensation) and recycled material at End of Life ( $Q_{Sout}/Q_p$ ; equal to 0.9 for generic -amorphous- PET from mechanical recycling of packaging).

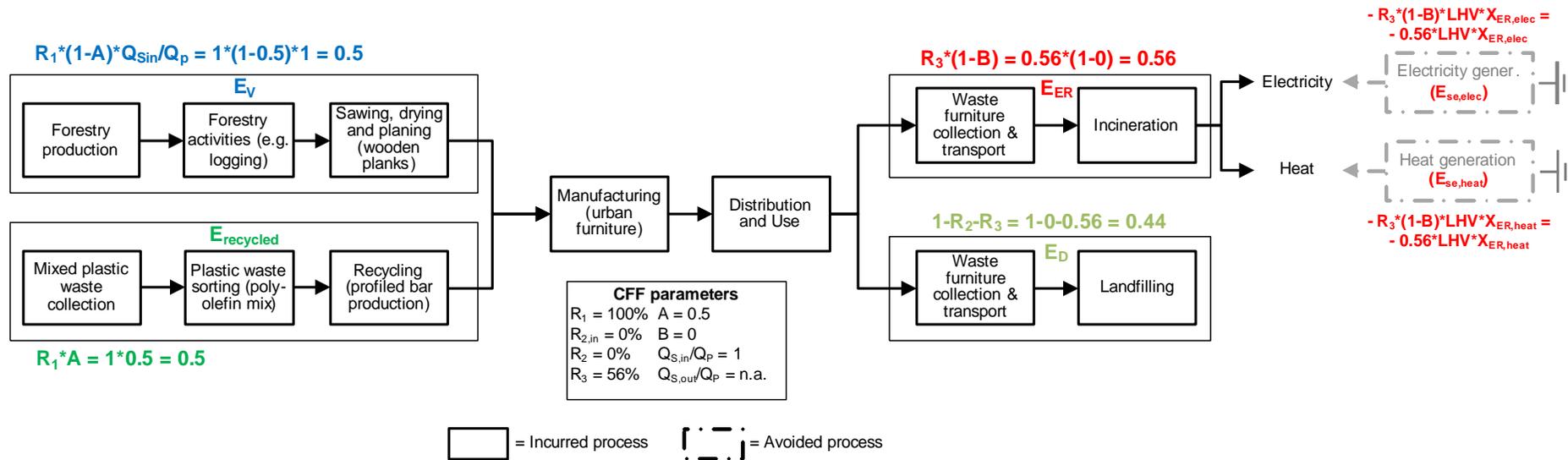
The example illustrated in Figure 13 refers to a hypothetical case of "open-loop" recycling, where mixed polyolefin waste (including e.g. PP, LDPE and PS waste) is recycled into extruded profiled bars used in the manufacturing of urban furniture (e.g. public park benches). Recycled plastic bars are here assumed to replace wooden planks traditionally used for public furniture, although it may also be the case that furniture relying on recycled plastic bars replace furniture made of concrete, virgin plastics or other materials, depending on the reference market/geography. Since in this example no mixing between primary material (wooden planks) and secondary material (plastic bars) is possible, the recycled content is necessarily set to 100% (supply-chain specific value appropriately evidenced by the company conducting the LCA study). Moreover, the A factor is assumed equal to 0.5 (no application- or material-specific default values are provided in Annex C), while in a real LCA study the relationship between supply and demand of recycled plastic bars shall be taken into account to determine the A factor, according to the rules specified in Section 4.4.10.2.2. Finally, the quality ratio  $Q_{Sin}/Q_p$  is set to 1, assuming that recycled plastic bars and wooden planks are used in equal volumes in furniture manufacturing, as long as both products are able to provide equivalent technical performances. This implies that any action performed during plastic

waste recycling into profiled bars to achieve comparable material performances is taken into account in the Life Cycle Inventory (e.g. use of additives and/or of an increased mass of plastic material per unit of final product). At End of Life, recycling is not considered a viable route in this example (i.e.  $R_2=0$ ; supply-chain specific value), with waste furniture being sent either to incineration (56%; i.e.  $R_3=0.56$ ) or landfilling (44%), based on the rates at which these End of Life options are applied to overall post-consumer plastic waste in Europe (as reported in EC, 2018a).



(\*) If the burdens of sorting are expressed per unit mass of input waste to be sorted, the same value (process share) calculated for collection and transport shall be applied in the modelling (i.e. 0.3). If such burdens are expressed per unit mass of sorted material leaving the plant, the calculation shall be adjusted to reflect the assumed sorting efficiency (91% in the example), corresponding to a share of:  $R_{2,in}^*0.91*(1-A) = 0.6*0.91*(1-0.5) = 0.27$ .

**Figure 12.** Example of application of the CFF to the life cycle of a disposable PET beverage bottle incorporating 30% recycled content and collected for recycling at End of Life at a rate of 60% (both rates being assumed as supply-chain specific values for which suitable evidence is provided by the company conducting the LCA study). Expressions reported above single processes or group of processes refer to relevant parts of the CFF and are used to calculate the share of burdens of each process or process group to be modelled in the Life Cycle Inventory.



**Figure 13.** Example of application of the CFF to the life cycle of a public urban furniture (e.g. park bench) made entirely of recycled plastic bars (from mixed polyolefin waste recycling). Expressions reported above single processes or group of processes refer to relevant parts of the CFF and are used to calculate the share of burdens of each process or process group to be modelled in the Life Cycle Inventory.

#### **4.4.10.3 Modelling of specific End of Life options**

A number of material/waste-specific or unspecific EF compliant LCI datasets are available for most of the commonly applied End of Life options, including mechanical recycling, industrial composting, incineration and landfilling. These datasets are accessible on the node <http://lcdn.thinkstep.com/Node/>, and can be used for free to develop a LCA study under a specific PEFCR.

As a general rule, material- and geography- specific inventory datasets shall be used to model specific End of Life options applied to the product in scope, or to other waste products/materials generated along the life cycle, in the geography of reference. In line with the datasets selection requirements reported in Section 4.6.3, the applied datasets shall be EF compliant or, in the absence of these, ILCD-Entry Level (EL) compliant (with the use of ILCD-EL compliant proxies to be reported in the "limitations" section of the LCA study report). If no material-specific datasets are available, more generic (EF compliant or ILCD-EL compliant) datasets for relevant material categories (e.g. for generic or average plastic waste rather than for waste of a specific polymer) shall be applied as proxies, provided that they are sufficiently representative of the specific materials to be treated or disposed of (and that their use as proxies is reported under the "limitations" section of the LCA study report). If no suitable datasets are available to be used as proxies, new EF compliant and material-specific inventory datasets shall be developed, based on the general modelling guidelines reported in Sections 4.4.10.5-4.4.10.11). If no EF compliant datasets can be developed (due to e.g. lack of suitable data), the process shall be excluded from the model, stating clearly this data gap in the "limitations" section of the LCA study report, to be validated by the verifier.

#### **4.4.10.4 Waste-specific parameters relevant for End of Life modelling**

A number of parameters related to the physical, chemical and biological characteristics of the product in scope (or of other waste product/materials generated throughout the product life cycle) are generally needed to properly develop new inventory datasets of the different end of life options they may undergo. These parameters include, for instance, the elemental composition of the product or of its material components, the proportion of each component, the density and energy content (lower heating value) of the product, as well as its mineralisation/biodegradation rate under specific (controlled) conditions (e.g. in composting or anaerobic digestion).

The physico-chemical properties considered in the modelling shall be specific for the product(s) in scope, reflecting those reported in the scope definition phase (Section 3.2.1). While elemental composition data referring to the product after use and collection as waste are preferable (and often applied in existing life cycle inventory datasets for waste treatment or disposal processes), the composition of the product as obtained from manufacturing is considered a suitable approximation, especially when innovative or emerging materials are assessed. In this case, composition data may also be derived from stoichiometric calculations.

The physical carbon content in (bio-based) products shall be considered for calculations related to the modelling of the End of Life stage, regardless of any allocation performed upstream in the product life cycle. For instance, if the (fossil) carbon content in waste PET is estimated to 0.625 kg C/kg PET, based on elemental composition analyses (or stoichiometry in the absence of more specific analytical values), such value shall be used for calculation (e.g. to quantify CO<sub>2</sub> emissions from incineration).

#### **4.4.10.5 Modelling of mechanical recycling processes**

In mechanical recycling, waste material is reclaimed to enable its use for manufacturing a new product. During mechanical recycling, plastic waste is for example ground, (further) separated from impurities or non-target materials via water-based density sorting/flotation, cleaned (e.g. washed in caustic solutions) and eventually recycled (i.e. normally regranulated into new polymer pellets). The quality of recycled material differs

depending on the quality of the input waste material and on the recycling processes applied. The quality of the input material in turns depends on the specific application(s) where the material is used, as well as by the applied waste collection system (e.g. dedicated take-back schemes, kerbside mono-material or multi-material collection, etc.) and the applied sorting technology.

Mechanical recycling may also be applied to specific bio-based plastic products, although currently this is mostly the case of products made of "drop-in" bio-based polymers (e.g. bio-based polyolefins such as bio-PET, bio-PE and bio-PP) which are chemically identical to their fossil-based counterparts. No established recycling pathways are currently in place for potentially recyclable bio-based plastic products made of alternative polymers, such as PLA. Most biodegradable polymers such as starch blends, PLA blends and bio-PBS are currently not recycled mechanically as they are mainly conceived to degrade under specific controlled conditions and hence are not suitable for recycling. Prerequisite for a valuable mechanical recycling of suitable bio-based plastic products is, as for conventional fossil-based plastics, a preferably source-separated (mono-material or multi-material) collection (depending on local conditions) and subsequent sorting into homogeneous polymer streams. For recyclable bio-based plastic products, the same collection schemes as those currently applied to conventional (fossil-based) plastics could be applied, provided that proper sorting into specific polymer streams is then performed. However, for products consisting of "drop-in" bio-based polymers no additional sorting is required as they can be mixed regardless of the feedstock used for production. On the other hand, collecting non-recyclable biodegradable plastics along with conventional plastics (from any feedstock) for further recycling may increase contamination of recycled conventional polymers with biodegradable material and ultimately affect their final quality, if no appropriate sorting is in place (e.g. in less advanced sorting facilities).

Material- and geography-specific inventory datasets shall be applied to model mechanical recycling of the product in scope, or of other waste products/materials generated along the life cycle, in the geography of reference. In line with the datasets selection requirements reported in Section 4.6.3, the selected datasets shall be EF compliant<sup>97</sup> or, in the absence of these, ILCD-Entry Level (EL) compliant (with the use of ILCD-EL compliant proxies to be reported in the "limitations" section of the LCA study report). If no material-specific datasets are available, more generic (EF compliant or ILCD-EL compliant) recycling datasets for relevant product/material categories (e.g. unspecified plastic waste rather than waste of a specific polymer) shall be applied as proxies, provided that the single unit operations modelled in the datasets are sufficiently representative for the recycling of the relevant products/materials (and that their use as proxies is reported under the "limitations" section of the LCA study report). If no suitable datasets are available to be used as proxies, new EF compliant and material-specific inventory datasets shall be developed based on the general modelling guidelines reported below, while relying on data collected from specific plants, the literature or other suitable data sources. If no EF compliant datasets can be developed (due to e.g. lack of suitable data), the process shall be excluded from the model, stating clearly this data gap in the "limitations" section of the LCA study report, to be validated by the verifier.

The key parameters required to develop the inventory of a specific mechanical recycling process are listed in Table 8. The modelling of mechanical recycling of the product in scope within the overall Life Cycle Inventory of the product shall instead be performed according to the Circular Footprint Formula (CFF). To apply the CFF, additional parameters related to the quality of the recycled material and of the replaced (primary) material are needed, as better specified in Section 4.4.10.2.5.

Recycling of bio-based products maintains the CO<sub>2</sub> taken up from the atmosphere during biomass growth within the recycled material until (after one or more recycling "loops") it

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<sup>97</sup> EF compliant datasets provided by the EC are available at <http://lcdn.thinkstep.com/Node/>, and can be used for free to develop a LCA study under a specific PEFCR (Product Environmental Footprint Category Rule).

ends up in other treatment or disposal options (e.g. incineration or landfilling) where it is totally or partially released back to the atmosphere.

**Table 8.** Requirements and recommendations on the main parameters and data needed to develop new inventory datasets for mechanical recycling of specific plastic waste products or materials.

<b>Parameter</b>	<b>Unit</b>	<b>Requirement / recommendation</b>
<i>Type of recycling technology(ies)</i>	-	Should reflect the relevant technology (or mix of technologies) applied in the geography and time period in scope
<i>Recycling efficiency</i>	% of input waste ultimately obtained as recycled material	Should be representative of the specific product or material targeted for recycling (with a preference for product-specific values)
<i>Energy demand - electricity-</i>	MJ/kg input waste	Should be based on process-specific consumption of the reference technology (or mix of technologies)
<i>Energy demand - thermal-</i>	MJ/kg input waste	
<i>Energy demand - mechanical- (e.g. fuel consumption)</i>	MJ/kg input waste	
<i>Water</i>	m <sup>3</sup> /kg input waste	Should be based on process-specific consumption of the reference technology (or mix of technologies)
<i>Ancillary materials (e.g. detergents, chemicals)</i>	kg/kg input waste	Should be based on process-specific consumption of the reference technology (or mix of technologies)
<i>Production of rejects (non-recycled material) - and respective fate-</i>	kg/kg input waste	Should be representative of the specific product or material targeted for recycling (with a preference for product-specific values)
<i>Wastewater production – and respective characteristics <sup>(1)</sup></i>	m <sup>3</sup> /kg input waste	Should be based on process-specific production of the reference technology (or mix of technologies)

<sup>(1)</sup> E.g. concentrations of main pollutants such as BOD, COD, TOC, Total Solids, TKN, NH<sub>4</sub><sup>-</sup>-N, NO<sub>3</sub><sup>-</sup>-N, NO<sub>2</sub><sup>-</sup>-N, PO<sub>4</sub><sup>3-</sup>-P, etc.

#### **4.4.10.6 Modelling of industrial composting processes**

Composting is a biological treatment process where biodegradable waste is typically converted, under aerobic conditions, into carbon dioxide, water, smaller amounts of methane and Non-Methane Volatile Organic Compounds (NMVOC), mineral salts of any other elements present in the waste, and into a residual solid fraction of simpler organic compounds (the compost), which is the main output of the process. Compost can serve as a soil amendment, maintaining soil carbon content and possibly replacing mineral fertilisers.

Composting can be distinguished between industrial composting and home composting. Industrial composting is a controlled process carried out in dedicated facilities. It typically consists of two main stages (i.e. oxidation followed by a maturation phase), where relevant process conditions (temperature, humidity, oxygen level, availability of microorganisms and residence time) are controlled. The composting performance is thus

generally stable and constantly ensured. Moreover, process emissions to air are normally controlled (e.g. through bio-filters in case of encapsulated systems) and any leachate is collected for treatment. A separation of non-composted residual materials (e.g. non-biodegradable plastic residues, films, etc.) is often performed as a final step. Home composting is a simple, one-stage and generally open-pile composting process where operating parameters are not controlled and can vary widely, resulting in a process where the average composting rate and performance are less predictable. Moreover, no emission control or final material separation is carried out in home composting. This section specifically focuses on controlled industrial composting, being home composting normally an unsuitable option for biodegradable plastic products.

Often, such as in the case of biodegradable plastic materials, the (bio)-degradation process starts with a chemical degradation (e.g. hydrolysis) of the previously shredded/fragmented material into simpler and smaller organic compounds (e.g. monomers). These are then subject to biodegradation (aerobic oxidation) by microorganisms available in the composting environment.

Prerequisite for considering industrial composting a viable treatment option for the product in scope is its compostability (including ultimate aerobic biodegradability) under controlled composting conditions. This can be assessed according to specific standards, such as EN 13432:2000 (for packaging products) and EN 14995:2006 (for plastic materials in general), and relevant testing methods specified in the latter.

Composting shall be modelled within the overall Life Cycle Inventory of the product according to the Circular Footprint Formula, setting the value of the A parameter to 0.5. When a new (EF compliant) inventory dataset needs to be developed for the composting process of suitable plastic products or materials, the modelling shall be carried out by reflecting, as far as possible, the specific scope of the LCA study in terms of analysed product/material, geography and reference period. In this respect, Table 9 provides a list of relevant parameters and data typically required for the modelling of industrial composting of biodegradable products and, where appropriate, specifies requirements or recommendations on how they shall/should be determined in a *Plastics LCA* study, as also partly discussed below.

The actual biodegradation rate (biodegradability) of the product in scope shall be considered, while the use of generic, product-unspecific data (e.g. for generic organic waste) shall be avoided. The biodegradation rate ("percentage of biodegradation") determined in accordance with the standardised testing methods (e.g. ISO 14855-1:2012)<sup>98</sup> recommended by the abovementioned European standards on compostability should be used, if available, unless more representative and substantiated values for real (full- or pilot-scale) composting conditions are available. Any alternative value applied shall be adequately documented, justified and submitted to the LCA study verification. If no specific data are available, a 90% biodegradation rate shall be considered as default value, according to the minimum biodegradability (percentage of biodegradation) required by such standards, which expresses the share of carbon in the digested product converted (mineralised) into CO<sub>2</sub>. If composting is considered as an end of life option, it is indeed expected that the product complies with the specified compostability standard requirements.

According with the interpretation of the mentioned compostability standards, the assumed (product-specific) biodegradation rate shall be considered as the rate at which carbon in the composted product is mineralised to CO<sub>2</sub> and CH<sub>4</sub> (i.e. as the carbon mineralisation rate), and shall be applied in the modelling accordingly (see below and Table 9 for more details). The same rate should also be applied to quantify (bio)-degradation of volatile solids in the product, unless more specific and representative data

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<sup>98</sup> The biodegradation rate (percentage of biodegradation) determined according to testing methods recommended in compostability standards refers to the percentage of carbon in the product/material that is converted into CO<sub>2</sub> (i.e. mineralised) during the biodegradation test. However, for modelling purposes, such mineralisation rate can also be considered a reasonable approximation of the overall percentage of biodegradation of the material/product (i.e. of the volatile solids contained in it).

are available in this respect (which if applied shall be properly justified and documented). Non-mineralised carbon that is not further emitted throughout the composting process (e.g. to water/leachate), shall be assumed to transfer to the residual organic material from composting of the product.

Biodegradation rates shall refer to the actual composition of the material used in the formulation/manufacturing of the product in scope, including any additives (which may affect the overall biodegradation rate). If additives are not or only partially biodegradable, the respective non-biodegraded portion shall be assumed to be entirely transferred to the residual organic material from composting, and their subsequent fate shall be properly modelled (i.e. as entirely emitted to soil if the residual material is applied on agricultural land). The same applies to any other non-biodegradable element or substance included in the product composition (e.g. metals).

Direct process emissions, production of residual composted material (if any) and its composition shall be determined taking into account the actual elemental composition of the product in scope sent to composting, and the assumed biodegradation rate. Suitable element-specific transfer coefficients (emission factors) shall be applied, where relevant, for the related calculations (e.g. those reported in Table 9). Emissions of substances which are not present in the product composition shall not be assigned to the composting process (e.g. nitrogen emissions shall not be considered if the nitrogen content of the product in scope is equal to zero). Similarly, no replacement of any mineral/synthetic fertilisers shall be modelled, if no nutrients that can be potentially transferred to residual organic material from composting are included in the composition of the product in scope (as it is currently the case for most biodegradable plastics).

**Table 9.** Requirements and recommendations on the main parameters and data needed to develop new inventory datasets for industrial composting of biodegradable plastic waste products or materials <sup>(1)</sup>.

<b>Parameter</b>	<b>Unit</b>	<b>Requirement / recommendation</b>
Type of composting technology(ies)	-	Should reflect the relevant technology (or mix of technologies) suitable for bioplastics composting applied in the geography and time period in scope
Biodegradation rate (carbon mineralisation rate to CO <sub>2</sub> /CH <sub>4</sub> )	% of C in the waste	Shall be product-specific and should be determined according with the testing methods recommended in compostability standards for plastics <sup>(2, 3)</sup> , unless more representative values for real (full- or pilot-scale) composting conditions are available <sup>(4)</sup> . Otherwise, a 90% biodegradation rate shall be applied (i.e. the minimum requirement from EN 13432/EN 14995)
Biodegradation rate (for VS in the product/material)	% of VS in the waste	Should be the same as the applied carbon mineralisation rate, unless more specific and representative values are available (these shall be properly justified and documented, if applied)
Production of residual composted material	kg/kg waste ww	Shall be consistent with the applied biodegradation rate for VS (e.g. Residual material = non-degraded VS + Initial Ash & Water content)
Leachate production (if any)	m <sup>3</sup> /kg waste ww	Should be based on process-specific production of the reference technology (or mix of technologies)
Characteristics and fertilising value of residual composted material (C, N, P, K and water content)	kg/kg DM	Shall be based on the specific composition of the product in scope, taking into account the applied biodegradation / mineralisation rate and the share of each element emitted to the environment during the composting process
<b>Energy demand (waste handling, aeration, etc.)</b>		
Electricity	MJ/kg waste ww	Should be based on the process-specific consumption of the reference technology (or mix of technologies)
Fuel (e.g. diesel)	m <sup>3</sup> /kg waste ww	
<b>Other inputs</b>		
Water consumption	m <sup>3</sup> /kg waste ww	Should be based on the process-specific consumption of the reference technology (or mix of technologies)
<b>Emissions to air <sup>(5)</sup></b>		
CO <sub>2</sub> (biogenic/fossil) <sup>(6)</sup>	kg/kg waste ww	99.99% of mineralised carbon in the product/material <sup>(7)</sup>
CH <sub>4</sub> (biogenic/fossil) <sup>(6)</sup>	kg/kg waste ww	0.01% of mineralised carbon in the product/material <sup>(7)</sup>

Parameter	Unit	Requirement / recommendation
NH <sub>3</sub>	kg/kg waste ww	98.5% of N content in the product/material <sup>(7)</sup>
N <sub>2</sub> O	kg/kg waste ww	1.4% of N content in the product/material <sup>(7)</sup>
N <sub>2</sub>	kg/kg waste ww	0.1% of N content in the product/material <sup>(7)</sup>
H <sub>2</sub> S	kg/kg waste ww	Should be based on process-specific emissions of the reference technology (or mix of technologies)
Terpenes		
NMVOC		
<b>Emissions to water or transfer to leachate (if any) <sup>(5,8)</sup></b>		
COD (Chemical Oxygen Demand)	kg/kg waste ww	Should be calculated as % of (non-mineralised) carbon in the product/material, or based on process-specific emissions of the reference technology (or mix of technologies)
NO <sub>3</sub> <sup>-</sup> -N	kg/kg waste ww	Should be calculated as % of N/P content (non-emitted to air) in the product/material, or based on process-specific emissions of the reference technology (or mix of technologies)
PO <sub>4</sub> <sup>3-</sup> -P	kg/kg waste ww	

<sup>(1)</sup> Not all the listed parameters may be relevant across all possible composting technologies.

<sup>(2)</sup> For instance EN 13432:2000 for plastic packaging and EN 14995:2006 for plastic materials in general.

<sup>(3)</sup> Shall reflect the composition of the material actually used in the formulation/manufacturing of the product in scope, including additives (which may affect the overall biodegradation rate).

<sup>(4)</sup> Any applied alternative value shall be adequately documented, justified and submitted to the LCA study verification.

<sup>(5)</sup> Appropriate elementary flows shall be selected to model emissions of specific substances to air and water, based on the list of flows reported in the latest version of the EF reference package available at the time of the study (currently 3.0).

<sup>(6)</sup> Depending on the origin of the carbon in the polymer.

<sup>(7)</sup> For enclosed tunnel composting facilities. Other composting technologies may apply different emission factors. These values should be applied for modelling unless more representative data are available for the reference technology.

<sup>(8)</sup> Net of any removal at wastewater treatment facilities.

#### **4.4.10.7 Modelling of anaerobic digestion processes**

Anaerobic digestion is a biological treatment process where biodegradable waste (or other feedstock) is converted, under anaerobic conditions, into biogas, water, and a residual fraction called digestate. Biogas is typically a mixture of Methane, Carbon Dioxide, NMVOC, N<sub>2</sub>, H<sub>2</sub>S and NH<sub>3</sub>, depending on the composition of the input waste. Carbon Dioxide and Methane are, however, the main components. Due to its high greenhouse gas potential, biogas needs to be properly managed in order to avoid its release to the atmosphere (e.g. during digestion, storage and combustion).

Energy may be recovered from the generated biogas, through combustion in cogeneration units, after removal of water vapour and acid gases. Biogas can also be upgraded to bio-methane for use as fuel for vehicles (alone or in combination with natural gas), or for distribution into the grid. The digestate may undergo a subsequent (aerobic) composting process, where it is converted to a more stabilised soil conditioner (in this case, the provisions on composting modelling reported in Section 4.4.10.6 applies also to composting of digestate). This is often the case when the feedstock consists of separately collected municipal organic waste. Alternatively, the digestate may be directly applied on land (as such or after dehydration), although this option unlikely applies to

biodegradable plastics since their degradation under anaerobic conditions is typically low or anyway partial (e.g. UBA, 2018), and an additional aerobic post-composting step of the residual material from digestion is needed to achieve suitable degradation levels. Abatement of air emissions in encapsulated digestion plants may be carried out through biofilters and/or scrubbers.

Prerequisite for considering anaerobic digestion a viable treatment option for the product in scope is at least its anaerobic treatability in a combined digestion process followed by aerobic stabilisation (composting) of the resulting digested material. The property can be currently assessed according to compostability standards covering also aspects relevant to any preceding anaerobic digestion phase, such as EN 13432:2000 (for packaging products) and EN 14995:2006 (for plastic materials in general). These standards provide requirements for minimum biodegradability (i.e. mineralisation) during the anaerobic digestion phase, and on other relevant aspects for the overall combined process (e.g. overall disintegration, quality of the resulting material after aerobic composting, etc.). However, these aspects are not specifically addressed for the case of anaerobic digestion applied alone as a single-stage process. Moreover, testing of biodegradability under anaerobic conditions is not mandatory according to such standards, when evaluating compostability of the product. Therefore, a product certified according to EN 13432:2000 or EN 14995:2006 is not necessarily suitable for processing within combined anaerobic-aerobic facilities (e.g. the product may remain intact or degrade/disintegrate to a very low extent). No dedicated standards specifically providing criteria and requirements for products suitable for anaerobic digestion are currently available.

Anaerobic digestion shall be modelled within the overall Life Cycle Inventory of the product according to the Circular Footprint Formula (e.g. under the “energy” part in the case of energy generated in the process). When a new (EF compliant) inventory dataset needs to be developed for anaerobic digestion of suitable plastic products or materials, it shall be modelled by reflecting, as far as possible, the specific scope of the LCA study in terms of analysed product/material, geography and reference period. In this respect, Table 10 provides a list of relevant parameters and data typically required for the modelling of industrial composting of biodegradable plastic products and, where appropriate, specifies requirements or recommendations on how they shall/should be determined in a *Plastics LCA* study, as also partly discussed below.

The actual biodegradation rate (biodegradability) of the product in scope under anaerobic digestion conditions shall be considered, as well as the corresponding biogas production. The use of generic, product-unspecific data (e.g. for generic organic waste) shall be avoided. The biodegradation rate (“percentage of biodegradation”) and corresponding biogas production determined according to the standardised testing methods (e.g. ISO 15985:2014 and ISO 14853:2016)<sup>99</sup> recommended by the abovementioned European standards (EN 13432:2000 or EN 14995:2006) should be used, if available, unless more representative and substantiated values for real (full- or pilot-scale) digestion conditions are available. Any alternative value applied shall be adequately documented, justified and submitted to the LCA study verification. If no specific data are available, a 50% biodegradation rate shall be considered as a default value, according to the minimum biodegradability (percentage of biodegradation) required by such standards, which expresses the share of anaerobically gasified carbon in the digested product. When values from standard testing methods (or the minimum 50% threshold) are applied, they should be reduced to account for the fact that they refer to experimental (laboratory) conditions, and may not be achieved in real, full-scale plants. The typical conversion yield of anaerobically biodegradable carbon in organic waste into biogas is 70% (as the average of the range 50-90%; Angelidaki and Batstone, 2010). Therefore, in the absence of more specific and representative data, this conversion factor should be applied also for biodegradable plastic products, which would imply considering an overall biodegradation

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<sup>99</sup> The biodegradation rate (percentage of biodegradation) determined according to testing methods recommended in compostability standards refers to the percentage of carbon in the product/material that is converted into CO<sub>2</sub> (i.e. mineralised) during the biodegradation test.

rate equal to 35% when the minimum 50% threshold from standards is assumed for the biodegradation rate.

According with the interpretation of the mentioned European standards, the assumed (product-specific) biodegradation rate shall be considered as the rate at which carbon in the digested product is mineralised to CO<sub>2</sub> and CH<sub>4</sub> (i.e. converted into biogas; carbon mineralisation rate), and shall be applied in the modelling accordingly (see below and Table 10 for more details). Non-mineralised carbon that is not further emitted throughout the digestion process (e.g. to water/leachate), shall be assumed to transfer to the residual organic material in the output digestate. Biodegradation of Volatile Solids (VS) in the product should be calculated considering a ratio between degraded VS and degraded Carbon equal to 1.89, unless more specific and representative values are available in this respect (which if applied shall be properly justified and documented).

Biodegradation rates shall refer to the actual composition of the material used in the formulation/manufacturing of the product in scope, including any additives (which may affect the overall biodegradation rate). If additives are not or only partially digestible, the respective non-biodegraded portion shall be assumed to be entirely transferred to the residual, non-digested or partially digested material output, and their subsequent fate shall be properly modelled (i.e. as entirely emitted to soil if the residual material is directly applied on agricultural land, or if it undergoes further composting but additives are not biodegradable). The same applies to any other non-biodegradable element or substance included in the product composition (e.g. metals).

Direct process emissions (e.g. fugitive Methane emissions), biogas composition, production of residual material in the output digestate, and its composition shall be determined taking into account the actual elemental composition of the product in scope sent to digestion, and the assumed biodegradation rate (see Table 10 for further details). Suitable element-specific transfer coefficients (emission factors) shall be applied, where relevant, for the related calculations. Waste-specific emissions of substances which are not present in the product composition shall not be assigned to the digestion process. For instance, no nitrogen emissions shall be modelled if the nitrogen content of the product in scope is equal to zero. Similarly, no fertilising value shall be assigned to the residual material in the output digestate, if no nutrients that can be transferred to it are included in the composition of the product in scope (as it is currently the case for most biodegradable plastics).

**Table 10.** Requirements and recommendations on the main parameters and data needed to develop new inventory datasets for anaerobic digestion of biodegradable plastic waste products or materials.

Parameter	Unit	Requirement / recommendation
Type of anaerobic digestion technology(ies)	-	Should reflect the relevant technology (or mix of technologies) suitable for bioplastics digestion applied in the geography and time period in scope
Biodegradation rate (carbon mineralisation rate to CO <sub>2</sub> /CH <sub>4</sub> )	% of C in the waste	Shall be product-specific and should be determined in accordance with the testing methods recommended in compostability/biodegradability standards for plastics ( <sup>1, 2</sup> ), unless more representative values for real (full- or pilot-scale) digestion conditions are available ( <sup>3</sup> ). Otherwise, a 50% biodegradation rate shall be considered ( <sup>4</sup> ). This default value and values from standard testing methods should be converted into a net biodegradation rate considering a conversion efficiency of anaerobically biodegradable carbon into biogas equal to 70% ( <sup>5</sup> )
Biodegradation rate (for VS in the product/material)	% of VS in the waste	Should be calculated considering a ratio $VS_{degraded}/C_{degraded} = 1.89$ , unless more specific and representative values are available (which shall be properly justified and documented)
Biogas production (total)	m <sub>n</sub> <sup>3</sup> CH <sub>4</sub> /kg waste ww	Shall be based on the carbon content in the product and the applied biodegradation rate
Biogas composition (CH <sub>4</sub> , CO <sub>2</sub> , N <sub>2</sub> , H <sub>2</sub> S, NH <sub>3</sub> , NMVOC)	%	Shall be based on the stoichiometry of the anaerobic degradation reaction and the C, H, O and N content in the product ( <sup>6</sup> )
Production of residual material in the output digestate	kg/kg waste ww	Shall be consistent with the applied carbon and VS biodegradation rate (e.g. Residual material = non-degraded VS + Initial Ash & Water content)
Leachate production (if any)	m <sup>3</sup> /kg waste ww	Should be based on process-specific production of the reference technology (or mix of technologies)
Characteristics and fertilising value of residual material in the output digestate (C, N, P, K and water content)	kg/kg DM	Shall be based on the specific composition of the product in scope, taking into account the applied biodegradation / mineralisation rate and the share of each element emitted to the environment during the digestion process
<b>Energy demand (waste handling, capture equipment, etc.)</b>		
Electricity	MJ/kg waste ww	Should be based on the process-specific consumption of the reference technology (or mix of technologies)
Thermal energy	MJ/kg waste ww	
Fuel (e.g. diesel)	m <sup>3</sup> /kg waste ww	
<b>Other inputs</b>		

<b>Parameter</b>	<b>Unit</b>	<b>Requirement / recommendation</b>
Water consumption	m <sup>3</sup> /kg waste ww	Should be based on the process-specific consumption of the reference technology (or mix of technologies)
<b>Energy recovery</b>		
Electricity generation	MJ/kg waste ww	Shall be based on the actual energy (i.e. Methane) content of the generated biogas and on the energy efficiency of utilisation units for the reference technology
Heat generation	MJ/kg waste ww	
<b>Direct emissions to air (e.g. fugitive emissions) (7)</b>		
CH <sub>4</sub> , CO <sub>2</sub> , N <sub>2</sub> O, NH <sub>3</sub> , etc.	kg/kg waste ww	Shall be consistent with the actual composition of the product in scope and the assumed biodegradation rate (no waste-specific emissions of substances excluded from the product composition shall be modelled)
<b>Emissions to water or transfer to leachate (if any) (7,8)</b>		
COD (Chemical Oxygen Demand)	kg/kg waste ww	Should be calculated as % of (non-mineralised) carbon in the product/material, or based on process-specific emissions of the reference technology (or mix of technologies)
N	kg/kg waste ww	Should be calculated as % of N/P content (non-emitted to air) in the product/material, or based on process-specific emissions of the reference technology (or mix of technologies)
P		

- (1) For instance EN 13432:2000 for plastic packaging and EN 14995:2006 for plastic materials in general.
- (2) Shall reflect the composition of the material actually used in the formulation of the final product, including additives (which may affect the overall biodegradation rate).
- (3) Any applied alternative value shall be adequately documented, justified and submitted to the LCA study verification.
- (4) According to the minimum biodegradability (biodegradation percentage) under anaerobic conditions required from EN 13432:2000 and EN 14995:2006 (i.e. 50%).
- (5) According to the typical conversion yield of anaerobically biodegradable carbon in organic waste into biogas (Angelidaki and Batstone, 2010), which should be applied unless more specific and representative data are available.
- (6) Limited to CH<sub>4</sub>, CO<sub>2</sub>, N<sub>2</sub> and NH<sub>3</sub>. For other components (e.g. NMVOC and H<sub>2</sub>S) typical average composition values for organic waste-derived biogas may be applied (where relevant).
- (7) Appropriate elementary flows shall be selected to model emissions of specific substances to air and water, based on the list of flows reported in the latest version of the EF reference package available at the time of the study (currently 3.0).
- (8) Net of any removal at wastewater treatment facilities.

#### **4.4.10.8 Modelling of composted or digested organic material use-on-land**

Residual organic material from biological treatment of biodegradable plastic products may be ultimately applied on agricultural land along with the (organic-waste derived) compost or digestate it is part of. However, direct on-land application of such residual material with digestate is currently of scarce relevance, since biodegradable plastic products typically show a (very) limited biodegradation under anaerobic digestion conditions (see e.g. UBA, 2018), so that additional aerobic treatment (post-composting) is generally needed. More relevant is hence the case of on-land application with compost, either from direct aerobic processing of the product, or from post-composting of the partially digested material from its anaerobic treatment.

In general, the base assumption for considering on-land application a viable option for residual material from biological treatment of bioplastic products is that such processes (i.e. composting or post-composting of anaerobically digested material) are properly operated, so that the input bio-plastic material is either mineralised (e.g. to CO<sub>2</sub>), metabolised by microorganisms, or converted into new biomass or simpler organic compounds. This means that no residues of non-biodegraded plastic material are present in the resulting organic material.

The use of materials/energy and emissions associated with the application on agricultural land of residual organic material from biological treatment of biodegradable plastic products should be quantified based on state-of-the-art literature data and emission factors estimated for the application of compost or digestate from organic waste (e.g. Yoshida et al., 2016 and Bruun et al., 2006). For the modelling of emissions, a 100-year time horizon after application should be considered. In this respect, Table 11 summarises the main parameters and data relevant to the modelling of on-land application and provides, where appropriate, requirements and recommendations on how they shall/should be determined, as also better discussed below. More representative data may be applied, if available, and shall be adequately documented and justified in the LCA study.

Diesel consumption for spreading operations was estimated at 0.00042 l/kg for compost and 0.00063 l/kg for digestate (Yoshida et al., 2016). The value estimated for compost should also be applied to residual organic material from composting of biodegradable plastic products or post-composting of the partially digested material from their anaerobic treatment (unless more representative data are available). The estimate provided for digestate should be applied, in the absence of better data, to residual material from anaerobic digestion of such products, wherever suitable biodegradation levels are achieved under anaerobic conditions, without the need of additional aerobic processing.

For compost derived from municipal organic waste, the mineralisation of carbon to CO<sub>2</sub>-C within 100 years from on-land application, simulated via agro-ecosystem modelling, equalled on average 89.3% of the carbon applied with compost (based on Bruun et al., 2006; average of values for all types of temperate soil). The average emission as CH<sub>4</sub>-C was 0.01% of the carbon applied, based on the same source. These values should be extended to organic material from composting or post-composting of biodegradable plastic products (when the type of soil where application occur is unknown), unless more representative data are available (which should be documented and justified). Carbon not mineralised within 100 years after application can be considered associated with stable organic compounds, and hence no longer released (back) to the atmosphere; no emissions should thus be modelled for carbon not converted to CO<sub>2</sub> or CH<sub>4</sub> over the first 100 years from application. Nitrogen emissions are normally not relevant for bioplastic-derived organic material, being the nitrogen content in bioplastics typically zero or very low. However, where relevant, average emissions to air of N<sub>2</sub>O-N and NH<sub>3</sub>-N from unspecified soils can be assumed as 3.24% and 1.61% of the N applied on-land with the residual material, respectively (based on Yoshida et al., 2016; average of values across all types of soil). Average emissions of NO<sub>3</sub>-N to water bodies can be estimated as 27% (16% to groundwater, 11% runoff to surface water) of the N applied on-land with the residual material, based on the same source and considering again average values across all types of soil. The reported emission factors for nitrogen should be applied in the modelling, unless more representative data are available (which shall be justified and documented).

For digestate, the mineralisation of carbon to CO<sub>2</sub>-C within 100 years equalled on average 87.5% of the carbon applied with the digested material (based on simulation results by Bruun et al. (2006) across all types of temperate soil). The average emission as CH<sub>4</sub>-C was 0.05% of the applied carbon, based on the same source. Average emissions of N<sub>2</sub>O-N and NH<sub>3</sub>-N to air from unspecified soils can be assumed, as 2.4% and 7.5% of the N applied on-land with the material, respectively (based on Yoshida et al.,

2016; average of values across all types of soil). Moreover, average emissions of NO<sub>3</sub>-N to water bodies can be quantified at 39.5% (20% to groundwater and 19.5% runoff to surface water) of the applied N, again for unspecified soils. The reported values should be applied to model emissions from on-land application of residual material from digestion of biodegradable plastic products (unless more representative data are available), wherever suitable biodegradation levels are achieved under anaerobic conditions, and direct application of digested material on-land is viable. Where this is not the case, and additional treatment of the partially digested material through an aerobic post-composting stage is needed, the emission factors reported above for on-land application of compost should be applied (unless more representative data are available, which shall be justified and documented).

No substitution of conventional NPK mineral/synthetic fertilisers shall be modelled for the use of residual organic material from (post)-composting or digestion of biodegradable plastic products, if these do not contain nutrients (NPK) that can and actually are transferred to the residual material (as is it is currently the case for most biodegradable plastics). Therefore, no avoided fertiliser production nor avoided emissions from their application shall be modelled.

Non-biodegradable elements in the product composition that are transferred to the residual material applied on-land, including any metals and additives, shall be assumed to be entirely emitted to soil, with their fate-exposure-effect accounted in underlying LCIA models in affected impact categories (e.g. toxicity-related categories). The same applies to any intermediate biodegradation product that is not ultimately mineralised to CO<sub>2</sub>/CH<sub>4</sub>, water and biomass. The fate of such substances and products may be alternatively modelled further at the inventory level (e.g. leaching to groundwater) if knowledge is available, considering emissions occurring in the relevant environmental compartment.

**Table 11.** Requirements and recommendations on the main parameters and data needed to model on-land application of residual organic material from composting or anaerobic digestion of biodegradable plastic waste products or materials <sup>(1)</sup>.

Parameter	Unit	Requirement / recommendation		
<b>Energy consumption (spreading operations)</b>				
Diesel (tractors)	m <sup>3</sup> /kg waste ww	Residual material from composting: 0.00042 l Residual material from digestion: 0.00063 l		
<b>Air emissions ( 2,3,4)</b>				
	<b>Type of soil</b>	<i>Sandy loam soil</i>	<i>Heavy clay soil</i>	<i>Coarse sand soil</i>
CO <sub>2</sub> -C (biogenic/fossil) <sup>(5)</sup>	% of C applied on land with residual organic material	88.68	89.13	90.11
		86.75	88.65	87.15
CH <sub>4</sub> -C (biogenic/fossil) <sup>(5)</sup>	% of C applied on land with residual organic material	0.01	0.01	0.01
		0.05	0.05	0.05
N <sub>2</sub> O-N	% of N applied on land with residual organic material	4.45	3.51	1.77
		2.78	2.47	1.89
NH <sub>3</sub> -N	% of N applied on land with residual organic material	1.61	1.61	1.61
		7.5	7.5	7.5
<b>Water emissions ( 2,3,4)</b>				
NO <sub>3</sub> <sup>-</sup> -N (to groundwater via leaching)	% of N applied on land with residual organic material	10.8	7.54	29.28
		17.77	11.6	30.85
NO <sub>3</sub> <sup>-</sup> -N (to surface water via runoff)	% of N applied on land with residual organic material	10.1	19.37	3.42
		19.58	33	5.86
<b>Soil emissions (4,6)</b>				
Metals (e.g. Cd, Cr, Cu, Hg, Ni, Pb, Zn) and other non-biodegradable elements (e.g. additives) <sup>(7)</sup>	kg/kg waste ww	Shall be equal to 100% of the amount applied on land with the residual material from composting or digestion		

<sup>(1)</sup> The reported values are based on the results by Yoshida et al. (2016; fuel consumption for spreading and nitrogen emissions) and Bruun et al. (2006, carbon emissions). Reported values should be applied unless more specific and representative data are available (which shall be justified and documented in the LCA study).

<sup>(2)</sup> Average emission factors across all types of soil reported may be applied if the specific soil type is unknown in the LCA study.

<sup>(3)</sup> For each emission factor, values in the first sub-row refer to application of residual organic material from composting, while those in the second one to application of residual organic material from anaerobic digestion.

<sup>(4)</sup> Appropriate elementary flows shall be selected to model emissions of specific substances to air and water and soil, based on the list of flows reported in the latest version of the EF reference package available at the time of the study (currently 3.0).

<sup>(5)</sup> Depending on the origin of carbon in the polymer (biogenic or fossil).

- (<sup>6</sup>) Or to other relevant environmental compartments if suitable data are available to model further the fate of substances initially released to soil.
- (<sup>7</sup>) Including any intermediate degradation product that is not ultimately mineralised to CO<sub>2</sub>/CH<sub>4</sub>, water and biomass.

#### **4.4.10.9 Modelling of in-situ biodegradation of plastic products**

In-situ biodegradation represents a possible End of Life option for specific agricultural plastic products, such as biodegradable mulching film, which after use is left on the soil and/or incorporated into it to allow biodegradation by microorganisms. Portions of non-biodegradable agricultural products (especially mulching film) may be also left on the field due to difficult removal conditions (e.g. after tearing of thinner films or due to inefficiency of collection equipment). In this case, in-situ degradation is not an appropriate and viable End of Life option, but rather represents a littering or waste mismanagement phenomenon, which is addressed in a separate section of this report (4.4.10.12).

In-situ biodegradation should be modelled considering a timeframe of 100 years after product use has terminated, consistently with the time horizon considered for on-land application of residual organic material from (post)-composting or anaerobic digestion (Section 4.4.10.7). The same time horizon is also normally applied to modelling of landfilling (Section 4.4.10.11). Table 12 summarises the main parameters and data relevant to the modelling of in-situ biodegradation of suitable plastic products and provides, where appropriate, requirements and recommendations on how they shall/should be determined, as also partly discussed below.

For biodegradable mulching film, a dedicated standard providing specific requirements and the corresponding testing methods is available (i.e. EN 17033:2018). For satisfactory biodegradation, a minimum biodegradation rate (percentage) equal to 90% has to be achieved over a maximum period of 24 months, when testing the material according to ISO 17556:2012. The biodegradation rate expresses the percentage of organic carbon in the material that is converted to CO<sub>2</sub> by the end of the test period (i.e. the carbon mineralisation rate). When modelling in-situ biodegradation of mulching film, the biodegradation rate (percentage) determined in accordance with the specified testing method (ISO 17556:2012) shall be considered, if quantified, unless more representative and substantiated values for real soil conditions or from improved simulation conditions (e.g. at larger scales) are available. Any alternative value applied shall be adequately documented, justified and submitted to the LCA study verification. If no specific data are available, a 90% biodegradation rate shall be considered as default value, according to the minimum rate required by EN 17033:2018). For products with no biodegradability standard of reference, the same minimum biodegradation (mineralisation) rate required for mulching film (i.e. 90%) should be applied as default, unless more representative and product-specific data from testing in real or properly simulated conditions are available (which shall be documented and justified).

Mineralised carbon can be reasonably assumed to be entirely converted into CO<sub>2</sub> and subsequently released (back) to the atmosphere. However, CH<sub>4</sub> production may also take place after the product is incorporated into the soil, where anaerobic conditions are more likely to occur. As default rule, mineralised carbon should be modelled as entirely converted to CO<sub>2</sub>, although alternative shares considering also conversion to CH<sub>4</sub> may be applied, as long as better knowledge is gained on biodegradation pathways of plastic products on/into the soil. Any alternative data shall be justified and documented, and submitted to the LCA study verification. Any carbon not mineralised within 100 years after use and beginning of the End of Life stage can be considered to be no longer released (back) to the atmosphere. No emissions should thus be modelled for carbon not converted to CO<sub>2</sub> (and CH<sub>4</sub>) over the first 100 years from the beginning of the End of Life stage.

Biodegradation pathways of plastic products and materials in soil are not yet completely understood. However, if all product/material components are ultimately biodegradable by

soil microorganisms (including any additives), and no hazardous or toxic substances are included in the product/material composition, the biodegradation process should ultimately lead to the sole formation of CO<sub>2</sub>/CH<sub>4</sub> (biogenic or fossil, depending on carbon origin in the polymer), water, mineral salts of any other element present in the product/material composition, and new soil biomass. On the other hand, non-degraded plastic fragments (including any micro-plastics), or any intermediate biodegradation products may be generated during degradation, and transferred to other environmental compartments or organisms before full biodegradation occurs. Such releases (both to soil or other relevant compartments) shall be properly taken into account in the inventory, as long as better knowledge is gained in this respect.

Any non-biodegradable elements present in the material composition, such as metals and non-biodegradable additives, shall be assumed to be entirely emitted to the soil during the 100-year time horizon considered for modelling, with their fate-exposure-effect accounted in underlying LCIA models in affected impact categories (e.g. toxicity-related categories). The same applies to any final or intermediate biodegradation product (beyond CO<sub>2</sub>, water, mineral salts and biomass) remaining in the soil at the end of such timeframe, including any intermediate compound from (partial) additive (bio)-degradation and any generated micro-plastic. The fate of such substances and products may be alternatively modelled further at the inventory level (e.g. leaching to groundwater) if knowledge is available, considering emissions occurring in the relevant environmental compartment.

**Table 12.** Requirements and recommendations on the main parameters needed to model in-situ biodegradation of biodegradable plastic waste products or materials.

Parameter	Unit	Requirement / recommendation
Biodegradation rate (carbon mineralisation rate)	% of C (and VS) in the waste	<p><u>For mulching film</u>: the percentage of biodegradation determined according to the testing method recommended in the standard EN 17033 shall be applied (if available), unless more representative values for real soil conditions or improved simulation conditions are available <sup>(1)</sup>. Otherwise, a 90% biodegradation rate shall be applied (according to the minimum requirement from EN 17033:2018)</p> <p><u>For products with no biodegradability standards of reference</u>: a 90% biodegradation rate should be assumed as default, unless more representative and product-specific data related to real or properly simulated conditions are available (which shall be documented and justified)</p>
<b><i>Air emissions (during biodegradation) <sup>(2)</sup></i></b>		
CO <sub>2</sub> (biogenic/fossil) <sup>(3)</sup>	kg/kg waste ww	100% of mineralised carbon in the product/material; should be assumed as default value if no better data are available on conversion rates to CO <sub>2</sub> /CH <sub>4</sub> <sup>(4)</sup>
CH <sub>4</sub> (biogenic/fossil) <sup>(3)</sup>	kg/kg waste ww	0% of mineralised carbon in the product/material; should be assumed as default value if no better data are available on conversion rates to CO <sub>2</sub> /CH <sub>4</sub> <sup>(4)</sup>
<b><i>Soil emissions <sup>(2,5)</sup> - during/at the end of biodegradation</i></b>		
Metals (e.g. Cd, Cr, Cu, Hg, Ni, Pb, Zn)	kg/kg waste ww	Shall be 100% of the amount of substance present in the product/material
Non-biodegradable additives <sup>(6)</sup>	kg/kg waste ww	Shall be 100% of the amount of substance contained in the product
Any final or intermediate biodegradation product <sup>(7)</sup> remaining in the soil after 100 years from the beginning of End of Life	kg/kg waste ww	Shall be calculated based on the product composition and biodegradation mechanism (as long as these are better understood)
<b><i>Other emissions to soil and/or other compartments (during biodegradation) <sup>(2)</sup></i></b>		
Non-degraded plastic fragments (including any micro-plastics)	kg/kg waste ww	Shall be calculated based on biodegradation mechanisms (as long as these are better understood)
Any intermediate biodegradation products	kg/kg waste ww	Shall be calculated based on the product composition and biodegradation mechanisms (as long as these are better understood)

<sup>(1)</sup> Any applied alternative value shall be adequately documented, justified and submitted to the LCA study verification.

<sup>(2)</sup> Appropriate elementary flows shall be selected to model emissions of specific substances to air and water and soil, based on the list of flows reported in the latest version of the EF reference package available at the time of the study (currently 3.0).

<sup>(3)</sup> Depending on the origin of carbon in the polymer.

- (<sup>4</sup>) The possibility that also CH<sub>4</sub> emissions occur once the product is incorporated into the soil shall be fully considered in the modelling and properly reflected in the inventory, as long as better knowledge is gained on biodegradation pathways of plastic products on/into the soil.
- (<sup>5</sup>) Or to other relevant environmental compartments if suitable data are available to model further the fate of substances initially released to soil.
- (<sup>6</sup>) Or any intermediate compound from their partial (bio)-degradation.
- (<sup>7</sup>) Beyond CO<sub>2</sub>, water, mineral salts and biomass.

#### **4.4.10.10 Modelling of incineration processes**

During incineration the waste product or material undergoes a combustion (oxidation) process, where it is fully or partially converted into a number of gaseous products such as CO<sub>2</sub>, water vapour, and other substances depending on the product/material composition. These may include, for instance, Sulphur Oxides (SO<sub>x</sub>) if the product/material contains sulphur, and Nitrogen Oxides (NO<sub>x</sub>), which originate both from nitrogen in the product/material and that present in combustion air. If the waste product/material also includes metals in its composition, these are also released in the combustion process, while metal wastes are normally mostly transferred to solid incineration residues (e.g. bottom ash and/or slag). Similarly, other inert wastes such as glass or ceramic products, or inert fractions or components of the waste material itself (e.g. sand, metal parts) end up in such residual fraction. Flue gases with combustion products are normally treated through a number of sequential cleaning steps, where air pollutants are removed with efficiencies depending on the type of air pollution control technologies applied. The energy content of the waste material (i.e. its lower heating value) is generally recovered as electricity, heat, or both (depending on the applied incineration technology).

Incineration of the product in scope at End of Life or of other waste products/materials generated along the life cycle of the product shall be modelled under the "energy" part of the Circular Footprint Formula. The modelling shall additionally reflect, as far as possible, the specific scope of the LCA study in terms of analysed product(s)<sup>100</sup>, geography and reference period.

Material- and geography-specific inventory datasets shall be applied to model incineration of the product in scope, or of other waste products/materials generated along the life cycle, in the geography of reference. In line with the datasets selection requirements reported in Section 4.6.3, the selected datasets shall be EF compliant<sup>101</sup> or, in the absence of these, ILCD-Entry Level (EL) compliant (with the use of ILCD-EL compliant proxies to be reported in the "limitations" section of the LCA study report). If no material-specific datasets are available, more generic (EF compliant or ILCD-EL compliant) incineration datasets for relevant material categories (e.g. average plastic waste rather than waste of a specific polymer) shall be applied as proxies, provided that they are sufficiently representative of the specific materials to be incinerated (and that their use as proxies is reported under the "limitations" section of the LCA study report). If no suitable datasets are available to be used as proxies, new EF compliant and material-specific inventory datasets shall be developed, based on the general modelling guidelines reported below. The development can be supported by dedicated modelling tools conforming to such principles. If no EF compliant datasets can be developed (due to e.g. lack of suitable product/material composition data), the process shall be excluded from the model, stating clearly this data gap in the "limitations" section of the LCA study report, to be validated by the verifier.

Any new (EF compliant) incineration inventory shall be developed by taking into account the specific chemical composition and energy content (Lower Heating Value; LHV) of the incinerated product (or of the respective material, if no product-specific data are available). These data shall be applied to determine, via element-specific transfer

<sup>100</sup> When incineration of the product in scope is modelled.

<sup>101</sup> EF compliant datasets provided by the EC are available at <http://lcdn.thinkstep.com/Node/>, and can be used for free to develop a LCA study under a specific PEFCR (Product Environmental Footprint Category Rule).

coefficients, direct process emissions, recovered energy, and more in general the distribution of each element in the waste product composition among the different process outputs (flue gas, ash/slag, and air pollution control residues). However, the ultimate emission of several substances (pollutants) that are subject to abatement (e.g. NO<sub>x</sub>, HCl, HF, SO<sub>2</sub>, particulate matter, some metals, dioxins and other organic compounds, etc.) depend on the respective concentration achievable in flue gas through the applied cleaning technologies, rather than on product composition. Similar considerations apply to intermediate oxidation products (e.g. CO and VOC), to substances originating (or used) during flue gas cleaning (e.g. NH<sub>3</sub>), and to N<sub>2</sub>O emissions. Such emissions should thus be modelled as process-specific, taking into account the average concentrations achieved in flue gas thanks to the average cleaning technology applied in the geography of reference, and the waste-specific flue gas production (m<sup>3</sup>/kg waste). Emissions related to substances that are not included in the product composition (and that cannot originate from other sources, e.g. thermal NO<sub>x</sub> originating from nitrogen in combustion air, or NH<sub>3</sub> used for cleaning) shall not be attributed, however, to the incineration process.

Technical parameters and assumptions applied in the modelling (e.g. energy efficiencies, configuration of the flue gas treatment line, related inputs and removal efficiencies, fate of process outputs, etc.) shall reflect the average technology or mix of technologies applied in the geography of reference.

Table 13 provides a list of relevant parameters and data typically required for incineration modelling and, where appropriate, specifies requirements or recommendations on how they shall/should be determined when developing a new incineration dataset according to the general provisions specified above.

**Table 13.** Requirements and recommendations on the main parameters and data needed to develop new inventory datasets for incineration of (plastic) waste products or materials.

Parameter	Unit	Requirement / recommendation
Type of incineration technology	-	Should reflect the relevant technology (or mix of technologies) applied in the geography and time period in scope
<b>Energy recovery</b>		
Electricity <sup>(1)</sup>	MJ/kg waste ww	Shall be based on the net energy content (LHV) of the product/material and on the net energy efficiencies (%LHV <sub>ww</sub> ) of the assumed reference technology (or mix of technologies) <sup>(2)</sup> .
Heat <sup>(1)</sup>	MJ/kg waste ww	
<b>Air emissions <sup>(3)</sup></b>		
CO <sub>2</sub> (fossil, biogenic) <sup>(4)</sup>	kg/kg waste ww	Shall be calculated based on the Carbon content of the product/material and on the respective origin (fossil or biogenic)
CO	kg/kg waste ww	Should be based on process-specific emissions (concentrations in flue gas) of the reference technology (or mix of technologies) and the waste-specific production of flue gas (m <sup>3</sup> /kg waste)
VOC		
NO <sub>x</sub> (thermal)		
N <sub>2</sub> O		
NH <sub>3</sub>		
Particles <sup>(5)</sup>		
Dioxin <sup>(6)</sup>	kg/kg waste ww	Should be based on process-specific emissions (concentrations in flue gas) of the reference technology, and the waste-specific production of flue gas (m <sup>3</sup> /kg waste).
HCl		
HF		
SO <sub>2</sub>	kg/kg waste ww	Shall be consistent with the product/material composition (i.e. no emission shall be accounted if the substance of origin is not included in the composition)
Metals (e.g. As, Cd, Cr, Hg, Ni, Pb, Zn)	kg/kg waste ww	Shall be based on the metal content in the product/material composition and the respective transfer coefficient to air <sup>(7)</sup>
<b>Production/composition of residues (and respective fate)</b>		
Bottom ash	kg/kg waste ww	Production should be based on the ash (or inert) and metal content of the product/material, or on the process-specific production of the reference technology (or mix of technologies) Composition should be based on transfer coefficients to bottom ash. Fate should be in line with the practices adopted in the reference geography

Parameter	Unit	Requirement / recommendation
Fly ash / Boiler ash	kg/kg waste ww	Production should be based on relevant composition parameters of the product/material, or on process-specific production of the reference technology (or mix of technologies).
Slag		Composition should be based on transfer coefficients to the specific output.  Fate should be in line with the practices adopted in the reference geography
<b>Ancillary inputs</b>		
Chemicals for air pollution control (e.g. Lime, Sodium Bicarbonate, Ammonia, Urea)	kg/kg waste ww	Should be based on the process-specific consumption of the reference technology (or mix of technologies)
Activated carbon		
Fuel (e.g. diesel)	kg/kg waste ww	

- (<sup>1</sup>) Amount exported from the incineration plant, net of any internal consumption.
- (<sup>2</sup>) Average gross energy efficiencies that should be currently applied for incineration at the EU level are 13.7% for electricity generation, and 31.8% for heat generation, unless more representative data are available.
- (<sup>3</sup>) Appropriate elementary flows shall be selected to model emissions of specific substances to air, based on the list of flows reported in the latest version of the EF reference package available at the time of the study (currently 3.0).
- (<sup>4</sup>) Depending on the origin of carbon in the incinerated product/material.
- (<sup>5</sup>) Considering appropriate elementary flows for the modelling, reflecting the actual size or size range of released particles, and to be selected among those reported in the latest version of the EF reference package available at the time of the study (currently 3.0). The following relevant flows are currently available: *Particles (PM0.2)*; *Particles (PM0.2 - PM2.5)*; *particles (PM2.5)*; *particles (PM2.5 - PM10)*; *particles (PM10)*; *particles (> PM10)*.
- (<sup>6</sup>) In any specific form/substance. Dioxin emissions shall be modelled using appropriate elementary flows, reflecting the actual substance released, and to be selected among those listed in the latest version of the EF reference package available at the time of the study (currently 3.0). The following relevant flows are currently available: *1,2,3,6,7,8-hexachlorodibenzo-p-dioxin*; *1,2,3,7,8,9-hexachlorodibenzo-p-dioxin*; *1,2,3,7,8-pentachlorodibenzo-p-dioxin*; *2,3,7,8-tetrachlorodibenzo-p-dioxin*; *2,3-dihydrothieno[3,4-b][1,4]dioxine*; *2,7-dichlorodibenzo-p-dioxin*; *dibenzo-p-dioxin*.
- (<sup>7</sup>) Emissions of some metals (e.g. As, Cd, Co, Cr, Ni and Pb) may also be modelled as being process-specific, taking into account their actual presence in the product/material composition (i.e. no emission shall be modelled if the specific metal is not part of the product composition).

#### **4.4.10.11 Modelling of landfilling**

A landfill is a site where waste products are buried in the ground. It can be either a managed (sanitary) or an unmanaged (unsanitary) landfill, although in Europe most landfills are of the managed type. In managed landfills, a number of technical measures are typically undertaken to prevent release of landfill gas and leachate to the environment, and to minimise the formation of leachate. Examples of such measures include the application of a bottom liner combined with the installation of a leachate collection system, subsequent treatment of collected leachate in a wastewater treatment plant prior to discharge to surface waterbodies, the application of a top (soil) cover, and the installation of a landfill gas capture system with flaring equipment. Energy recovery from landfill gas is also generally performed in managed landfills.

In landfills, a mix of inert, aerobic and anaerobic conditions can be found, so that both aerobic and anaerobic biodegradation of the waste product/material may take place depending on its characteristics (biodegradability), on site-specific landfill conditions

(temperature, rainfall, level of coverage, microbial activity) and on the applied landfilling technology (e.g. technical measures to stimulate biodegradation). Biodegradation and decomposition processes lead to the generation of landfill gas, which mainly consists of methane and carbon dioxide. As the landfill gas moves upwards through above waste and cover layers, methane can be partly oxidised to carbon dioxide. Some decomposition products are also transferred to leachate originating via rainwater infiltration through the landfill body, and ultimately emitted to surface water (if leachate is collected and treated, and until the capture system is in place) or leached to groundwater. If landfill gas is captured, it can be either used for energy recovery (e.g. in a gas engine) or flared (to at least oxidise methane to carbon dioxide). Both pathways generate combustion products that are fully or partly released as air emissions. Captured leachate can be routed to dedicated or existing wastewater treatment plants for cleaning, before being released to surface waterbodies. Additional environmental burdens from landfilling are associated with fuel and electricity supply to run on-site operations.

Landfilling of the product in scope and End of Life or of other waste products/materials generated along the life cycle of the product shall be modelled under the “disposal” part of the Circular Footprint Formula. The modelling shall additionally reflect, as far as possible, the specific scope of the LCA study in terms of analysed product(s)<sup>102</sup>, geography and reference period.

Material- and geography-specific inventory datasets shall be applied to model landfilling of the product in scope, or of other waste products/materials generated along the life cycle, in the geography of reference. In line with the datasets selection requirements reported in Section 4.6.3, the selected datasets shall be EF compliant<sup>103</sup> or, in the absence of these, ILCD-Entry Level (EL) compliant (with the use of ILCD-EL compliant proxies to be reported in the “limitations” section of the LCA study report). If no material-specific datasets are available, more generic (EF compliant or ILCD-EL compliant) datasets for relevant material categories (e.g. average plastic waste rather than waste of a specific polymer) shall be applied as proxies, provided that they are sufficiently representative of the specific materials to be landfilled (and that their use as proxies is reported under the “limitations” section of the LCA study report). If no suitable datasets are available to be used as proxies, new EF compliant and material-specific inventory datasets shall be developed, based on the general modelling guidelines reported below. The development can be supported by dedicated modelling tools conforming to such principles. If no EF compliant datasets can be developed (due to e.g. lack of suitable product/material composition data), the process shall be excluded from the model, stating clearly this data gap in the “limitations” section of the LCA study report, to be validated by the verifier.

Any new (EF compliant) landfilling inventory shall be developed by taking into account the specific chemical composition and (bio)-degradation rate (e.g. decomposition and/or mineralisation rate) of the landfilled product (or of the respective material, if no product-specific data are available). These data shall be used to determine, via element-specific transfer coefficients, the distribution of degradation products originating during landfilling between the generated landfill gas and leachate, as well as to estimate ultimate emission of these substances to air and waterbodies. Emissions of substances originating from elements not included in the composition of the input waste shall not be modelled in the inventory. For biodegradation (decomposition/mineralisation) rates over the 100-year timeframe that should be considered for modelling (see below), values determined according to available standard testing methods (e.g. ASTM D5526 – 18<sup>104</sup> or other appropriate standardised tests) should be used, if available. Alternatively, proper values from the literature may be applied, which shall be adequately documented and justified.

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<sup>102</sup> When landfilling of the product in scope is modelled.

<sup>103</sup> EF compliant datasets provided by the EC are available at <http://lcdn.thinkstep.com/Node/>, and can be used for free to develop a LCA study under a specific PEFCR (Product Environmental Footprint Category Rule).

<sup>104</sup> ASTM D5526-18 – Standard Test Method for Determining Anaerobic Biodegradation of Plastic Materials Under Accelerated Landfill Conditions.

Default (bio)-degradation/mineralisation rates for selected polymers are reported in Table 15, and should be applied in the modelling wherever more representative product-specific data are not available (which if applied shall be adequately documented and justified).

Technical parameters and assumptions relevant to the modelling (e.g. landfill gas capture rate, utilisation rate of captured gas, energy efficiency of gas engines, lifetime of the leachate capture system, etc.) shall reflect the average technology (or mix of technologies) applied in the geography of reference. Relevant meteorological parameters (e.g. mean annual precipitation, temperature and evapotranspiration) shall also be representative of the geography in scope. Table 14 provides a list of relevant parameters and data typically required for landfilling modelling and, where appropriate, specifies requirements or recommendations on how they shall/should be determined when developing a new landfilling dataset according to the general provisions specified above and further discussion below on the time horizon applied in modelling.

A time horizon of 100 years after waste deposition is typically considered when modelling waste degradation/decomposition in landfills and the resulting emissions with biogas and leachate, to develop waste-specific inventory datasets (e.g. Doka, 2009; Kupfer et al., 2020). This is mainly because, in managed municipal solid waste landfills, intense biological activity comes to an end within approximately the first 80 years after deposition (with a duration of the actual methane/active phase up to 30 years from deposition; Doka, 2009). Mixed municipal solid waste is hence mostly (bio)-degraded during such a timeframe. As a consequence, laboratory tests conducted to determine material (bio)-degradability under landfill conditions (e.g. Accelerated Landfill Conditions tests) are normally operated to simulate a period of 100 years from disposal. However, the situation may be different for unmanaged landfills (where biodegradation can extend for longer periods), and for specific (plastic) products differing from average municipal solid waste, which may further (bio)-degrade even beyond the simulated 100-year period. This is the case, for instance, of certain biodegradable plastic materials, which showed a still ongoing biodegradation at the end of the test, such as reported in Vermeulen (2007). Moreover, over an infinite time horizon (which is typically looked at in LCA) additional (bio)-degradation may take place for the product (to an extent depending on the material type). While the use of a different (longer) time horizon is not recommended in this method, for reasons of consistency with traditional LCA practice, known situations of incomplete biodegradation during the simulated 100 years of landfilling shall be properly taken into account in the interpretation of the LCA results, wherever these are significantly affected by the applied 100-year horizon.

The share of landfilled product or material that is not (bio)-degraded within 100 years from deposition can be assumed to undergo no further degradation, and carbon not mineralised within such a timeframe can be considered to be no longer released (back) to the atmosphere. No emissions should thus be modelled for carbon not released as CO<sub>2</sub> or CH<sub>4</sub> over the first 100 years from application. The amount of non-released biogenic and fossil carbon (per functional unit) may be reported as "additional environmental information" (see Section 3.2.5). However, this information shall be complemented by specific considerations on whether the applied (bio)-degradation rates properly reflect the long-term degradation potential of the product, or if further degradation (and hence carbon release) can be expected even beyond the applied 100-year time horizon (according to the discussion reported above).

For products relying on non-biodegradable polymers such as fossil-based and bio-based PET, PE and PP, as well as ABS, PC, etc., a mineralisation rate equal to 1%<sup>105</sup> of the carbon originally disposed of with the material should be considered within a 100-year time horizon. For products relying on biodegradable polymers (e.g. PLA, Starch-based polymer, PBS) the material-specific mineralisation rates reported in Table 15 should be

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<sup>105</sup> This value is in line with the overall degradation rate applied by Doka (2009) for conventional, non-biodegradable polymers such as PET, PE and PP, which can be extended to the respective drop-in bio-based variants.

applied. Alternative, more representative and product-specific rates may be applied for both biodegradable and non-biodegradable polymers, if available. These shall be adequately documented in the LCA study report and justified.

Due to storage mechanisms in the landfill body, substances originating from product decomposition over the first 100 years from disposal may actually be released (to leachate only) even after such a timeframe. This is, for instance, the case of metals liberated from a decomposed waste matrix and then re-precipitated in secondary solid phases, leading to delayed (or long-term) emissions of such substances (after 100 years from deposition). Delayed emissions shall not be inventoried when developing landfilling datasets conforming to the present method<sup>106</sup>.

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<sup>106</sup> Note also that characterisation factors of long-term emissions of toxic substances are set to zero in the default impact assessment models for Human Toxicity and Ecotoxicity impact categories.

**Table 14.** Requirements and recommendation on the main parameters and data needed to develop new inventory datasets for landfilling of (plastic) waste products or materials.

Parameter	Unit	Requirement / recommendation
Type of landfill technology	-	Should reflect the relevant technology (or mix of technologies) applied in the geography and time period in scope
Time horizon (for emission modelling)	Years	Should be 100 years (long-term emissions beyond 100 years shall not be inventoried)
(Bio)-degradation (decomposition) rate (Carbon mineralisation rate) in 100 years	% of waste  (% of C in the waste)	Shall be product-specific (or material-specific) and preferably determined according to standardised testing methods (e.g. ASTM D5526 – 18; Accelerated Landfill Conditions test)  Default values reported in Table 15 for selected polymers should be applied, unless more representative product-specific data are available <sup>(1)</sup>
<b>Technological parameters</b>		
Landfill height / area	m / m <sup>2</sup>	Should reflect the characteristics of the average technology (or mix of technologies) applied in the geography and time period in scope
Capture rate of landfill gas (active phase)	% of generated landfill gas	
Utilisation rate of captured landfill gas	% of captured landfill gas	
Energy efficiencies in landfill gas utilisation	% of LHV <sub>landfill gas</sub>	
Oxidation of CH <sub>4</sub> in non-captured landfill gas (surface layers)	% of CH <sub>4</sub> in landfill gas	
Leachate capture efficiency (and duration of capture)	% of generated leachate  (years)	
<b>Meteorological parameters</b>		
Mean annual precipitation	mm/year	Should reflect the average meteorological conditions of the geography in scope
Mean annual temperature	°C	
Mean annual actual evapotranspiration	mm/year	
<b>Landfill gas generation</b>		
Landfill gas production	m <sub>n</sub> <sup>3</sup> /kg waste ww	Shall be based on the actual composition of the landfilled product/material and on specific transfer

<b>Parameter</b>	<b>Unit</b>	<b>Requirement / recommendation</b>
Landfill gas composition (CH <sub>4</sub> , CO <sub>2</sub> , CO, NMVOC, NO <sub>2</sub> , particles, etc.)	% of landfill gas	coefficients of decomposition products into landfill gas
<b>Leachate generation</b>		
Leachate production	m <sup>3</sup> /kg waste ww	Shall be based on average infiltration rate (mm/m <sup>2</sup> y) of the geography and technology in scope, and on landfill site characteristics (height, waste density, residence time of leachate etc.) of the relevant technology
Leachate composition (e.g. COD, NH <sub>4</sub> <sup>+</sup> , NO <sub>3</sub> <sup>-</sup> , PO <sub>4</sub> <sup>3-</sup> , metals, etc.)	%	Shall be based on the actual composition of the landfilled product/material and specific transfer coefficients of decomposition products to landfill leachate
<b>Energy demand (waste handling, equipment operation, etc.)</b>		
Electricity	MJ/kg waste ww	Should be based on the process-specific consumption of the reference technology (or mix of technologies)
Thermal energy	MJ/kg waste ww	
Fuel (e.g. diesel for vehicles)	m <sup>3</sup> /kg waste ww	
<b>Energy recovery</b>		
Electricity	MJ/kg waste ww	Shall be based on the energy content (LHV) of landfill gas generated from the specific product/material landfilled and on process-specific energy efficiencies (%LHV) of the reference technology
Heat	MJ/kg waste ww	
<b>Material demand (landfill infrastructure and operation)</b>		
Lining materials (e.g. polyethylene)	kg/kg waste ww	Should be based on estimated requirements of the applied technology (or mix of technologies) in the reference geography
Pipes for leachate collection (PE or PVC)	kg/kg waste ww	
Cover material (e.g. soil and clay)	kg/kg waste ww	

(<sup>1</sup>) Any applied alternative value shall be adequately documented, justified, and submitted to the LCA study verification.

**Table 15.** Default (bio)-degradation (carbon mineralisation) rates for selected biodegradable and non-biodegradable polymers over the first 100 years from disposal in a managed (sanitary) landfill for municipal solid waste <sup>(1)</sup>.

Polymer	(Bio)-degradability / C mineralisation rate (%)	Source
Conventional non-biodegradable polymers <sup>(2)</sup> : PET, PE, PP, PS, EPS, PUR, ABS, PC, PVC, etc.	1	Doka (2009)
PLA	1	Kolstad et al. (2012)
TPS blends	11 <sup>(3)</sup>	Vermeulen (2007)
PLA blends	11	As TPS blends <sup>(4)</sup>
PBS / Bio-PBS	1	As conventional non-biodegradable polymers (Doka, 2009) <sup>(5)</sup>
PEF	1	As conventional non-biodegradable polymers (Doka, 2009)

<sup>(1)</sup> The reported values should be applied in the modelling of landfilling unless more representative product-specific data are available (which shall be adequately documented and justified).

<sup>(2)</sup> Both fossil-based and bio-based, as far as relevant for the specific polymer.

<sup>(3)</sup> This value is observed at the end of an Accelerated Landfill Conditions test where biodegradation was still ongoing (i.e. the biodegradation curve had not yet reached a plateau). Therefore, a higher biodegradation rate can be actually expected over a 100-year timeframe or beyond.

<sup>(4)</sup> In the current absence of more representative data, which should be applied if available.

<sup>(5)</sup> In the current absence of specific data for landfilling and considering the very low biodegradability under anaerobic digestion conditions (UBA, 2018). More representative, product-specific data should be applied if available.

#### **4.4.10.12 Release of plastic products in the environment (including product litter)**

At present, no sufficient knowledge and data are available to model the burdens (and potential impacts) of plastic products released as macro-plastics into the environment (e.g. from product littering and waste mismanagement) in a reliable and accurate manner. Beyond the release of the product itself, such burdens may also include, for instance, the subsequent release of additives, generation of micro- and nano-plastics from fragmentation, or any other organic compound from degradation (be it biodegradation, oxidation/reduction, or light-degradation). Degradation or biodegradation pathways of plastic products in the environment still have to be better investigated and understood, so that no specific requirements or recommendations can be currently given in this method on the modelling of the potential burdens resulting from their release. However, the MarILCA project<sup>107</sup> is currently ongoing as a joint effort from the UN Life Cycle Initiative and the Forum for Sustainability through Life Cycle Innovation, which aims at integrating potential environmental impacts of marine litter, especially plastic, in LCA. The outcome of this project may hence be useful in the future to also provide specific rules on the modelling of environmental burdens (i.e. emissions) from the release of plastic products into the environment at the inventory level.

The contribution of the product(s) in scope to macro-plastics generation and release into the terrestrial, riverine and/or marine environment should be currently estimated and reported as “additional environmental information” in the LCA study (Section 3.2.5). First methodological guidelines and data have been recently developed based on a multi-stakeholder initiative (the *Plastic Leak Project* – PLP) to estimate the release of macro-plastics at End of Life due to product littering by consumers and waste mismanagement (Peano et al., 2020). The method is well documented and complemented by supporting

<sup>107</sup> Described at <https://marilca.org/>.

data, and may be applied to calculate additional environmental information until more refined methods, possibly building on larger consensus and improved data, become available. Other accounting approaches have also been recently developed (as briefly described in the overview available in Annex I), although they are in most cases not based on consensus and/or not yet sufficiently solid or complete to be recommended (and hence should not be currently applied to provide additional environmental information).

Beyond macro-plastics generation and release at End of Life, the *PLP method* also addresses micro-plastics generation and release from different sources throughout the upstream product life cycle (e.g. due to pellet losses from processes and tyre abrasion). It may hence also be applied to provide an estimate of the potential release of micro-plastics from the product supply chain, which should be included as well in the LCA study as “additional environmental information” (Section 3.2.5).

An operational description of the *PLP method* is reported in Annex I, including the main equations, parameters and default values applicable to estimate the release of macro-plastics and micro-plastics throughout the product life cycle. While building upon state-of-the-art knowledge on macro- and micro-plastics generation, default values provided for the parameters of the PLP are affected by current knowledge gaps, scarcity of measured data and uncertainty (e.g. in relation to actual leakage/loss rates). Alternative values to default ones may hence be applied, for instance to reflect any more recent data and evidence, or to account for more site/region-specific and/or supply-chain specific data. Such alternative values shall be adequately documented and justified in the LCA study report and submitted to the verification process.

#### **4.4.11 Extended product lifetime**

Extending a product lifetime, due to reuse or refurbishment of the product, may result into two situations:

1. Reuse/refurbishment into a product with original product specifications (providing the same function).

In this situation (situation 1), the product lifetime is extended into a product with the original product specifications (providing the same function) and shall be included in the FU and reference flow. The user of this method shall describe how reuse or refurbishment is included in the calculations of the reference flow and in the overall life cycle model, taking into account the “how long” of the FU.

2. Reuse/refurbishment into a product with different product specifications (providing another function), or reuse of a same identical product for another function.

In this situation (situation 2), the reuse/refurbishment of a product results into a product with different product specifications (providing another function) or in the use of a same product for another function (e.g. reusing shopping bags for household waste collection). This situation shall be considered as part of the CFF, as a form of recycling (see Section 4.4.10.2). Also, old parts that have been changed during refurbishment shall be modelled under the CFF.

##### **4.4.11.1 Reuse rates (situation 1)**

The reuse rate is the number of times a material or packaging is used at the factory. This is often also called trip rates, reuse time or number of rotations. This may be expressed as the absolute number of reuses or as % of reuse rate. For example, a reuse rate of 80% equals 5 reuses. Equation 4 describes the conversion:

$$\text{Number of reuses} = \frac{1}{100\% - \% \text{ reuse rate}} \quad [\text{Equation 4}]$$

The number of reuses applied here refers to the total number of uses during the life of the material or packaging. It includes both the first use and all the following reuses.

#### **4.4.11.2 How to apply and model the 'reuse rate' (situation 1)**

The number of times a material is reused affects the environmental profile of the product at different life cycle stages. The following five steps explain how the different life cycle stages shall be modelled with reusable materials, using packaging as an example:

1. *Material Acquisition*: the reuse rate determines the quantity of packaging material consumed per product sold. The material consumption shall be calculated by dividing the actual weight of the packaging by the number of times this packaging is reused. For example, a 1l glass bottle weights 600 grams and is reused 10 times (reuse rate of 90%). The material use per litre is 60 gram (= 600 gram per bottle / 10 reuses).
2. *Transport from packaging manufacturer to the product factory* (where the products are packed): the reuse rate determines the quantity of transport that is needed per product sold. The transport impact shall be calculated by dividing the one-way trip impact by the number of times this packaging is reused.
3. *Transport from product factory to final client and back*: additionally to the transport needed to go to the client, the return transport shall also be taken into account. To model the total transport, Section 4.4.7 on modelling transport and logistics shall be followed.
4. *At product factory*: once the empty packaging is returned to the product factory, energy and resource use shall be accounted for cleaning, repairing or refilling (if applicable).
5. *Packaging End of Life*: the reuse rate determines the quantity of packaging material (per product sold) to be treated at End of Life. The amount of packaging treated at End of Life shall be calculated by dividing the actual weight of the packaging by the number of times this packaging was reused.

#### **4.4.11.3 Packaging reuse rates**

A packaging return system can be organized by:

1. The company owning the packaging material (company-owned pools), or
2. A third party, e.g. the government or a pooler (third party operated pools).

This may have an influence on the lifetime of the material as well as the data source to be used. Therefore, it is important to separate these two return systems.

**For company-owned packaging pools** the reuse rate shall be calculated using supply chain-specific data. Depending on the data available within the company, two different calculation approaches may be used (see Options a and b presented below). Returnable glass bottles are used as an example, but the calculations also apply for other types of company-owned reusable packaging.

**Option a:** use of supply chain-specific data, based on accumulated experience over the lifetime of the previous glass bottle pool. This is the most accurate way to calculate the reuse rate of bottles for the previous bottle pool and is a proper estimate for the current bottle pool. The following supply chain-specific data is collected:

- Number of bottles filled during the lifetime of the bottle pool (#Fi);
- Number of bottles at initial stock plus purchased over the lifetime of the bottle pool (#B).

The following parameters are then determined:

$$\text{Reuse rate of the bottle pool} = \frac{\#F_i}{\#B} \quad [\text{Equation 5}]$$

$$\text{The net glass use} \left( \frac{\text{kg glass}}{\text{l beverage}} \right) = \frac{\#B \times (\text{kg glass/bottle})}{\#F_i} \quad [\text{Equation 6}]$$

This calculation option shall be used:

1. With data of the previous bottle pool when the previous and current bottle pools are comparable. This implies having the same product category, similar bottle characteristics (e.g. size), comparable return systems (e.g. way of collection, same consumer group and outlet channels), etc.;
2. With data of the current bottle pool when future estimations/extrapolations are available on: (i) the bottle purchases, (ii) the volumes sold, and (iii) the lifetime of the bottle pool.

The data shall be supply chain-specific and shall be verified by an external verification, including the reasoning for choosing option "a" rather than option "b" described below.

**Option b:** If no real data is tracked, the calculation shall be done partly based on assumptions. This option is less accurate due to the assumptions made and therefore conservative/safe estimates shall be used. The following data is needed:

- Average number of rotations of a single bottle, during one calendar year (if not broken). One loop consists of filling, delivery, use, and back to filler for washing (#Rot);
- Estimated lifetime of the bottle pool (LT, in years);
- Average percentage of loss per rotation. This refers to the sum of losses at consumer and the bottles rejected at filling sites (%Los).

$$\text{Reuse rate of the bottle pool} = \frac{LT}{(LT \times \%Los) + \left( \frac{1}{\#Rot} \right)} \quad [\text{Equation 7}]$$

This calculation option shall be used when option "a" is not applicable (e.g., the previous pool is not usable as reference). The data used shall be verified by an external verification, including the reasoning for choosing option "a" rather than option "b" described below.

#### **4.4.11.4 Average reuse rates for company-owned pools**

LCA studies that have company-owned reusable packaging pools in scope shall use company-specific reuse rates, calculated following the rules specified in Section 4.4.11.3.

#### **4.4.11.5 Average reuse rates for third party operated pools**

The following reuse rates shall be used in LCA studies that have third party operated reusable packaging pools in scope, unless data of better quality are available:

- Glass bottles: 30 trips for beer and water, 5 trips for wine<sup>108</sup>;
- Plastic crates for bottles: 30 trips<sup>109</sup>;

<sup>108</sup> Assumption based on the monopoly system of Finland:  
<http://ec.europa.eu/environment/waste/studies/packaging/finland.pdf>

<sup>109</sup> Technical approximation as no data source could be found. Technical specifications guarantee a lifetime of 10 years. A return of 3 times per year (between 2 to 4) is taken as first approximation.

- Plastic pallets: 50 trips (Nederlands Instituut voor Bouwbiologie en Ecologie, 2014)<sup>110</sup>;
- Wooden pallets: 25 trips (Nederlands Instituut voor Bouwbiologie en Ecologie, 2014)<sup>111</sup>.

Alternative values may be used in the study, but shall be justified and the data source shall be provided. If a specific packaging is not covered in the list above, sector-specific data shall be collected and applied. New values shall be subject to the LCA study verification, if applicable.

The user of this method shall indicate if company-owned or third party operated pools were in scope, and which calculation method or default reuse rates were used.

#### **4.4.12 Electricity use**

Electricity use from processes and activities occurring throughout the product life cycle and included in the system boundary shall be modelled according to the provisions specified in Section 4.4.2 of the suggestions for updating the PEF method reported in Zampori and Pant (2019), dealing with electricity use. However, further clarification is here provided on how to model the use of secondary/recovered energy (e.g. electricity recovered from waste incineration facilities) as process input or as a commodity (Section 4.4.12.1).

##### **4.4.12.1 Use of secondary energy and fuels**

Secondary or recovered energy of any form, including secondary materials used as fuels, shall be modelled according to the Circular Footprint Formula analogously to the use of secondary material input, while replacing the A factor with the B factor. The B factor is set to 0 as default, meaning that secondary/recovered energy shall be modelled as 100% primary energy (considering the primary energy sources(s) assumed to be replaced from recovered energy). This also applies to the use of secondary fuels for direct heating in industrial processes.

#### **4.4.13 Capital goods - infrastructures and equipment**

Capital goods (including infrastructures) and their End of Life should be excluded<sup>112</sup>, unless there is evidence from previous studies that they are relevant. If capital goods are included, the LCA study report shall include a clear and extensive explanation, reporting all assumptions made and details of performed calculations.

As a general rule, the modelling of capital goods shall be based on linear depreciation, i.e. the respective environmental burdens shall be evenly distributed throughout the useful life of the good, and/or the related production/processing amount over such period, as applicable. The expected service life of capital goods shall be taken into account as the useful life (and not the duration to evolve to an economic book value of 0).

#### **4.4.14 Sampling procedure**

In some cases, a sampling procedure is needed to limit the collection of company-specific data only to a representative sample of plants, farms etc. For instance, a sampling procedure may be needed when multiple production sites are involved in the production of the same product, in case the same raw material/input material comes from multiple sites, or in case the same process is outsourced to more than one subcontractor/supplier.

<sup>110</sup> The less conservative number is used.

<sup>111</sup> Half of plastic pallets is used as approximation.

<sup>112</sup> This provision is based on the experience gained in the development of existing PEFCRs and the related supporting studies (nearly 70 studies for a total of 20 PEFCRs), showing a limited contribution of infrastructures and equipment to the overall potential impacts.

The user of this method shall: (i) specify in the LCA study report if sampling was applied, (ii) follow the general requirements described in this section, and (iii) indicate which approach was chosen.

Different procedures exist to derive a representative sample. Users of this method shall derive the representative sample via a stratified sample, i.e. a sample ensuring that sub-populations (strata) of a given population are each adequately represented within the whole sample of the study.

Using a stratified sample will always achieve greater precision than a simple random sample, provided that the sub-populations have been chosen so that the items of the same sub-population are as similar as possible in terms of the characteristics of interest. In addition, a stratified sample guarantees better coverage of the population<sup>113</sup>.

The following procedure shall be applied in order to select a representative sample as a stratified sample:

1. Define the population;
2. Define homogenous sub-populations (stratification);
3. Define the sub-samples at sub-population level;
4. Define the sample for the population starting from the definition of sub-samples at sub-population level.

For further guidance and examples on each step, the reader is referred to Section 4.4.6 of the suggestions for updating the PEF method reported in Zampori and Pant (2019), addressing the same sampling procedure.

#### **4.4.15 Greenhouse gas emissions and removals**

Three main categories of greenhouse gas (GHG) emissions and removals shall be distinguished, each contributing to a specific sub-category of the impact category 'Climate Change':

1. Fossil GHG emissions and removals (contributing to the sub-category 'Climate Change – fossil');
2. Biogenic carbon emissions and removals (contributing to the sub-category 'Climate Change – biogenic');
3. Carbon emissions from land use and land use change (contributing to the sub-category 'Climate Change – land use and land use change').

Effects (e.g. credits) associated with temporary and permanent carbon storage and/or delayed carbon emissions shall not be considered in the calculation of the Climate Change impact indicator (see also Section 4.4.15.4). This means that all emissions and removals occurring over the life time of the product and the first 100 years after its End of Life through final disposal/treatment<sup>114</sup> shall be accounted for as emitted at the same point in time (i.e. "now") and there is no discounting of emissions or removals over time (in line with ISO 14067:2018).

The sub-categories 'Climate Change – fossil', 'Climate Change – biogenic' and 'Climate Change – land use and land use change', shall be reported separately if they show a contribution of more than 5% each to the total Climate Change impact score<sup>115</sup>.

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<sup>113</sup> The practitioner has control over the sub-populations that are included in the sample, whereas simple random sampling does not guarantee that sub-populations (strata) of a given population are each adequately represented within the final sample. However, one main disadvantage of stratified sampling is that it may be difficult to identify appropriate sub-populations for a population.

<sup>114</sup> Including landfilling, incineration, composting/digestion followed by on-land application, and in-situ (bio)-degradation.

<sup>115</sup> For example, if 'Climate Change – biogenic' contributes with 7% to the total Climate Change impact and 'Climate Change – land use and land use change' contributes with 3%, only the former contribution

#### **4.4.15.1 Sub-category 1: Climate Change – fossil**

This sub-category covers GHG (carbon) emissions to any environmental media originating from the oxidation and/or reduction of fossil fuels by means of their transformation or degradation (e.g. combustion, composting, digestion, landfilling, etc.). This category also includes emissions from peat mineralisation (i.e. from peat once extracted from nature, and not in-situ emissions, which are part of the 'land use and land use change' category, see Section 4.4.15.3), and emissions/uptakes due to calcination/carbonation of limestone.

Fossil CO<sub>2</sub> uptake and the corresponding emissions (e.g. due to carbonation) shall be modelled in a simplified way (i.e., no emissions or uptakes shall be modelled). When the amount of fossil CO<sub>2</sub> uptake is required for calculating additional environmental information, the CO<sub>2</sub> uptake may be modelled with the flow "CO<sub>2</sub> (fossil), uptake from air".

*Modelling requirements:* The flows falling under this sub-category shall be modelled consistently with the list of elementary flows in the most recent version of the EF reference package available at the time of the study (currently 3.0)<sup>116</sup>. The names ending with '(fossil)' (e.g., 'Carbon dioxide (fossil)' and 'Methane (fossil)') shall be used.

#### **4.4.15.2 Sub-category 2: Climate Change – biogenic**

This sub-category covers carbon emissions to air (CO<sub>2</sub>, CO and CH<sub>4</sub>) originating from the oxidation and/or reduction of aboveground biomass by means of its transformation or degradation (e.g. combustion, composting, digestion, landfilling), as well as CO<sub>2</sub> uptake from the atmosphere through photosynthesis during biomass growth<sup>117</sup>. Carbon exchanges from native forests<sup>118</sup> shall be modelled under sub-category 3 (including connected soil emissions, derived products or residues).

*Modelling requirements:* The flows falling under this definition shall be modelled consistently with the list of elementary flows in the most recent version of the EF reference package available at the time of the study<sup>119</sup>, and using the flow names ending with '(biogenic)', e.g. 'Carbon dioxide (biogenic)' and 'Methane (biogenic)'. A mass allocation shall be applied to model the biogenic carbon flows.

All biogenic carbon emissions and removals shall be modelled in the Life Cycle Inventory of a plastic product incorporating bio-based material, including biogenic CO<sub>2</sub> uptakes and releases. The modelling should especially ensure the application of correct data for uptake and releases related to the biogenic carbon content in the product, to avoid distortions from potential inconsistencies in the carbon balance/inventory when any emissions from feedstock processing or CO<sub>2</sub> flows related to any co-product, by-product or residue/waste cannot be properly or completely tracked throughout the life cycle. Note, however, that the default characterisation factors for biogenic CO<sub>2</sub> uptake and releases are set to zero (0) in this method, fully conforming to the most recent version of the EF reference package, so that any unintended imbalance between carbon uptakes and releases modelled in the inventory does not ultimately compromise the LCA results for the Climate Change impact category. It is also acknowledged that a commonly accepted approach to handle biogenic carbon emissions and removals from products is currently missing, and consensus has not yet been achieved in the scientific community. This is also highlighted by the variety of approaches applied in the LCA studies analysed

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('Climate Change – biogenic') shall be reported, along with the total value of the Climate Change impact and with the contribution of 'Climate Change – fossil'.

<sup>116</sup> <https://epca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>.

<sup>117</sup> CO<sub>2</sub> uptake from the atmosphere contributes to define the carbon content of products, biofuels and aboveground plant residues such as litter and dead wood.

<sup>118</sup> Native forests – represents native or long-term, non-degraded forests. Definition adapted from table 8 in Annex V C(2010)3751 to Directive 2009/28/EC. In principle, this definition excludes short-term forests, degraded forests, managed forests, and forests with short-term or long-term rotations.

<sup>119</sup> <https://epca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>.

in the preliminary review reported in Annex K (and particularly in Sections K.2.2.1 and K.3).

For intermediate products such as polymers or unspecified plastic parts (cradle-to-gate studies), the biogenic carbon content at factory gate (physical content) shall always be reported as 'additional technical information', along with the non-biogenic (e.g. fossil) carbon content.

#### **4.4.15.3 Sub-category 3: Climate Change – land use and land use change (LULUC)**

This sub-category accounts for carbon uptakes and emissions (CO<sub>2</sub>, CO and CH<sub>4</sub>), as well as other GHG emissions (e.g. N<sub>2</sub>O), originating from carbon stock changes caused by land use and land use change. This sub-category includes biogenic carbon exchanges from deforestation, road construction or other soil activities (including soil carbon emissions). Carbon emissions from peat when left in its place (i.e. in situ) are also included (e.g. from converting peat areas to plantations). All GHG emissions related to the conversion of native forests are included and modelled under this sub-category (including connected emissions of GHG from the soil, products derived from native forest<sup>120</sup> and residues), while their CO<sub>2</sub> uptake is excluded.

A distinction is made between direct and indirect land use change. Direct land use change (dLUC) occurs as the result of a transformation from one land use type into another, which takes place in a unique land cover, possibly incurring changes in the carbon stock of that specific land, but not leading to a change in other systems. Examples of direct land use change are the conversion from forestland to cropland, or of land used for growing crops to industrial use. Emissions from direct land use change shall be considered in the LCA study, according to the modelling requirements specified below.

Indirect land use change (iLUC) occurs when a certain (direct) change in land use, or in the use of the feedstock grown on a given piece of land, induces changes in land use outside the system boundary, i.e. in other land use types. While indirect land use change is addressed in this method, it shall not be taken into account in the calculation of the environmental profile of the studied product, due to the lack of an agreed method to be applied for quantification. However, its effects should be evaluated and reported as additional environmental information, following the modelling recommendations provided below.

Annex J provides additional background information and discussion on land use changes, including broader definitions of both dLUC and iLUC, as well as an overview of models currently available to quantify GHG emissions from iLUC in a LCA context.

*Modelling requirements:* The flows falling under this definition shall be modelled consistently with the list of elementary flows in the most recent version of the EF reference package<sup>121</sup> available at the time of the study, and using the flow names ending with '(land use change)'. Biogenic carbon uptakes and emissions shall be inventoried separately through the appropriate elementary flows.

For **direct land use change (dLUC)**: all carbon emissions and removals from dLUC shall be modelled following the modelling guidelines of PAS 2050:2011 (BSI, 2011) and the supplementary document PAS 2050-1:2012 (BSI, 2012) for horticultural products. The provisions given below on the modelling of changes in soil carbon stock shall also be taken into account, as far as appropriate, in the modelling of direct land use change.

In PAS 2050:2011 (Appendix C), two main types of previous land use are considered for the modelling of dLUC: i) transformation from grassland (to annual or perennial crop) and ii) transformation from forest land (to annual or perennial crop). Quoting (with minimum adjustments) PAS 2050:2011 (BSI, 2011), the following provisions are given.

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<sup>120</sup> Following the instantaneous oxidation approach in IPCC 2013 (Chapter 2).

<sup>121</sup> <https://epca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>.

"Large emissions of GHGs can result as a consequence of land use change. Removals as a direct result of land use change (and not as a result of long-term management practices) do not usually occur, although it is recognized that this could happen in specific circumstances. [...]. All forms of land use change that result in emissions or removals shall be included. [...].

The GHG emissions and removals arising from direct land use change shall be assessed and quantified for any input to the life cycle of a product originating from land and shall be included in the assessment. The emissions arising from the product shall be assessed on the basis of the default land use change values provided in PAS 2050:2011 Annex C, unless better data is available. For countries and/or land use change types not included in this annex, the emissions arising from the product shall be assessed considering the GHG emissions and removals occurring as a result of direct land use change in accordance with the relevant sections of the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006; Volume 4, Chapter 5, Section 5.3, or later editions, as available).

The assessment of the impact of land use change shall include all direct land use changes occurring not more than 20 years, or a single harvest period, prior to undertaking the assessment (whichever is the longer). The total GHG emissions and removals arising from direct land use change over the period shall be included in the quantification of GHG emissions of products arising from this land on the basis of equal allocation to each year of the period<sup>122</sup>.

Where it can be demonstrated that the land use change occurred more than 20 years prior to the assessment being carried out, no emissions from land use change should be included in the assessment, i.e. dLUC should be set to zero (no dLUC occurs). Where the timing of land use change cannot be demonstrated to be more than 20 years, or a single harvest period, prior to making the assessment (whichever is the longer), it shall be assumed that the land use change occurred on 1<sup>st</sup> January of either:

- The earliest year in which it can be demonstrated that the land use change had occurred; or
- On 1<sup>st</sup> January of the year in which the assessment of GHG emissions and removals is being carried out for the product in scope.

The following hierarchy shall apply when determining the GHG emissions and removals arising from land use change occurring not more than 20 years or a single harvest period prior to making the assessment (whichever is the longer):

1. Where the country of production is known and the previous land use is known, the GHG emissions and removals arising from land use change shall be those resulting from the change in land use from the previous land use to the current land use in that country (additional guidelines on the calculation can be found in PAS 2050-1:2012);
2. Where the country of production is known, but the former land use is not known, the GHG emissions arising from land use change shall be the estimate of the average emissions from the land use change for the specific crop investigated in that country (additional guidelines on the calculation can be found in PAS 2050-1:2012);
3. Where neither the country of production nor the former land use is known, the GHG emissions arising from land use change shall be the weighted average of the average land use change emissions of the specific commodity investigated in the countries in which it is grown.

Knowledge of the prior land use can be demonstrated using a number of sources of information, such as satellite imagery and land survey data. Where records are not

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<sup>122</sup> In case of variability of production over the years, a mass allocation should be applied.

available, local knowledge of prior land use can be used. Countries in which a crop is grown can be determined from import statistics, and a cut-off threshold of not less than 90% of the weight of imports may be applied. Data sources, location and timing of land use change associated with inputs to products shall be reported.”

Note that when options #2 and #3 in the list above are applied, interpretation of results shall be done with care, as the modelled land use changes and related emissions may not be relevant for the specific product in scope and the related supply-chain. In particular, any PEF CR developed based on this method should specify the suitability of options #2 and #3 when results are to be used for comparisons or comparative assertions.

Intermediate products such as polymers (cradle-to-gate studies) derived from native forest feedstocks shall always report as meta-data (in the ‘additional technical information’ section of the LCA study report): (i) their carbon content (physical content and allocated content) and (ii) that corresponding carbon emissions shall be modelled with ‘(land use change)’ elementary flows.

Biogenic carbon in wood extracted from native forest and in any derived products is to be treated like fossil carbon, same as peat.

For **soil carbon stock**: carbon emissions from changes in soil carbon stock shall be included and modelled under this sub-category (e.g. from rice fields). Soil carbon emissions derived from aboveground residues (except from native forest) shall be modelled under sub-category 2, such as the application of non-native forest residues or straw. Soil carbon uptake (accumulation) shall be excluded from the LCA results, as it is highly questionable whether a mid-term uptake (beyond 100 years) or a truly long-term one (>10,000 years) can be guaranteed in practice. This includes, for example, carbon uptake from grasslands or from improved land management through tilling techniques or other management actions taken in relation to agricultural land. Soil carbon storage may only be included in the LCA study as additional environmental information and if proof is provided. If legislation has different modelling requirements for the sector, such as the EU Decision on greenhouse gas accounting from 2013 (Decision 529/2013/EU)<sup>123</sup>, which indicates carbon stock accounting, it shall be modelled according to the relevant legislation and provided under additional environmental information. Note, however, that overall long-term uptake and emissions of soil carbon due to direct land use change are both consistently accounted in the quantification of the respective net GHG emissions, considering net variations in soil carbon stock between the initial and final stages of both the previous and resulting land use<sup>124</sup>.

For **indirect land use change (iLUC)**: iLUC GHG emissions from land clearing and the resulting contribution to the Climate Change indicator should be quantified by applying the iLUC GHG emission factors proposed in the EU 2015/1513 Directive, annex V and VIII (EC, 2015). These iLUC factors were originally reported in the Directive per type of crop possibly used as a feedstock (differing between starch-rich, sugar-rich, and oil-rich) as gCO<sub>2</sub>-eq. MJ<sup>-1</sup>, and were intended to be applied to the assessment of energy-rich products, e.g. biofuels. In order to apply these figures to non-energy products (e.g. plastic products), it is necessary to recalculate these factors as kg CO<sub>2</sub>-eq. ha<sup>-1</sup> a<sup>-1</sup>. This may be done by applying typical energy yields for each individual crop-type (MJ ha<sup>-1</sup> a<sup>-1</sup>) based, for instance, on the figures reported in a recent study by Valin et al. (2015). The results are reported in Table 16.

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<sup>123</sup> Decision No 529/2013/EU of the European Parliament and of the Council of 21 May 2013 on accounting rules on greenhouse gas emissions and removals resulting from activities relating to land use, land-use change and forestry and on information concerning actions relating to those activities. Accessible at <https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex:32013D0529>.

<sup>124</sup> This means that, for instance, even if the resulting land use after transformation leads initially to a relevant soil carbon loss, any subsequent uptake until the final stage of such land use will be accounted for. However, any possible interim uptake of additional carbon via specific measures (e.g. “carbon-enriching techniques” such as no or minimum tillage) is excluded, as measurements of soil carbon content over a limited period are insufficient to capture the actual long-term effect in a reliable manner. Long-term experiments are needed for this purpose, but currently there are only few examples, and the information basis is hence not sufficiently broad to enable inclusion of soil carbon accumulation from such practices.

The iLUC contribution of the product in scope to the Climate Change indicator can be quantified according to Equation 8, i.e. by individually multiplying the total amount of agricultural (arable and pasture) land demanded by each crop (*i*) used as feedstock for the product ( $ha \cdot a \cdot FU^{-1}$ ), with the appropriate iLUC GHG emission factor in Table 16 (last row), and then aggregating the resulting contribution of each crop. The amount of land demanded by each crop used as feedstock for the bio-based plastic product has to be calculated from the Life Cycle Inventory based on the total specific agricultural land demand of the crop (e.g.  $ha \cdot a \cdot kg_{crop}^{-1}$ ), the specific crop input to polymer production ( $kg_{crop} \cdot kg_{polymer}^{-1}$ ) and the amount of polymer needed per FU ( $kg_{polymer} \cdot FU^{-1}$ ).

$$= \sum_i \text{iLUC contribution to Climate Change } [kgCO_2 - eq \cdot FU^{-1}] \\ = \sum_i \text{total land demand}_i [ha \cdot a \cdot FU^{-1}] \cdot \text{iLUC GHG emission factor}_i [kgCO_2 \cdot ha^{-1} \cdot a^{-1}]$$

[Equation 8]

**Table 16.** iLUC GHG emission factors that should be applied to quantify the iLUC contribution of the product in scope to the Climate Change indicator (recalculated on the basis of the figures in the EU Directive 2015/1513 (EC, 2015)).

Parameter	Unit	Crop type		
		Starch-rich	Sugar-rich	Oil-rich
iLUC GHG emission factor (energy basis)	gCO <sub>2</sub> -eq. MJ <sup>-1</sup>	12	13	55
Yield	MJ ha <sup>-1</sup> a <sup>-1</sup>	51000	135000	37000
iLUC GHG emission factor <sup>(1)</sup> (land basis)	kgCO <sub>2</sub> -eq. ha <sup>-1</sup> a <sup>-1</sup>	612	1755	2035

<sup>(1)</sup> If required for calculation, non-amortised factors expressed as kgCO<sub>2</sub>-eq. ha<sup>-1</sup> can be calculated considering an amortisation time equal to 20 years.

Alternative models and/or factors may be additionally applied to calculate GHG emissions from iLUC and the resulting contribution to the Climate Change indicator, such as the model presented in Schmidt et al. (2015; briefly described in Annex J with an operational perspective). However, the results obtained applying any alternative model or factor shall not replace those calculated based on the factors reported in Table 16, and shall be presented separately under additional environmental information.

#### 4.4.15.4 Temporary carbon storage and delayed emissions

Temporary carbon storage takes place when (biogenic) carbon removed from the atmosphere during biomass growth is embodied in a (bio-based) product and is stored in it for a limited amount of time. A consequence of this storage are delayed emissions, i.e. emissions that are released over time (e.g. during/after long-use or final disposal stages), compared to an instantaneous emission at a specific time *t* closer to uptake<sup>125</sup>. For instance, let us assume the fictional case of a wood-derived plastic furniture with an estimated life span of 60 years (starting from the year of harvest/production), at the end

<sup>125</sup> Delayed emissions may equally occur also for fossil-based carbon in products, when embodied carbon is released over time through long-use or disposal stages, rather than as a single emission at time *t* closer to extraction and product manufacturing.

of which it is disposed of through incineration. The CO<sub>2</sub> is taken up by the plant used for the production of the furniture, is stored in the furniture itself for 60 years, and is released only when the furniture is incinerated at its End of Life. Related CO<sub>2</sub> emissions are thus delayed by 60 years with respect to the year of harvest/production, as they occur only at the end of the product life span rather than at a very early stage from uptake, which could be the case if that wood was instead harvested and used for energy purposes. If a time horizon of e.g. 100 years from harvest/production is chosen to evaluate the effects (potential impact) of GHGs once released in the environment, delayed CO<sub>2</sub> emissions from incineration would hence contribute to such an impact only for a period of 40 years, and not over the full 100-year time frame (as it is the case of emissions occurring at the beginning of the product life cycle). However, while GHG (and particularly CO<sub>2</sub>) emissions related to the product-C degradation can be timed applying so-called dynamic accounting/assessments (inherently accounting for the effects of temporary carbon storage and delayed emissions) these approaches imply selecting appropriate time horizons for calculation. This has the ultimate effect of incurring the problem of identifying an appropriate time horizon (e.g. 20y, 50y, 100y, 1000y). Different time horizons would be associated to different results of the dynamic assessment. This issue has been largely discussed in the literature (e.g., among the others, Brandão et al., 2013, Guest et al., 2013 and Lévassieur et al., 2016), but no consensus has been achieved so far and no commonly accepted approach is currently available. Moreover, delayed emissions would continue to contribute to the impact on climate also beyond the applied time horizon (unless very long periods are considered) and hence any results calculated on this basis would need to be interpreted carefully and properly communicated. On the other hand, the application of dynamic accounting approaches may be relevant in policy context to be developed under the Paris agreement, which aims at mitigation measures to limit the global temperature rise to the target year 2100, and hence where even a delayed radiative forcing in the mid-term (due to e.g. temporary carbon storage) may play a role.

The effects of temporary storage of biogenic carbon and/or delayed carbon emissions shall not be considered in the Life Cycle Inventory nor in the calculation of the Climate Change impact indicator (in line with ISO 14067:2018). This means that all emissions and removals occurring over the product life cycle and within 100 years from final disposal/treatment<sup>126</sup> at End of Life shall be accounted for as emitted at the same point in time (t=0), regardless of their distance from uptake (or from incorporation of carbon into the product). Moreover, no discounting of emissions/removals over time shall be performed, neither at the inventory nor at the impact assessment level.

#### **4.4.16 Offsets**

The term “offset” is frequently used with reference to third-party greenhouse gas mitigation activities, e.g. regulated schemes in the framework of the Kyoto Protocol (CDM – Clean Development Mechanism, JI – Joint Implementation, ETS - Emissions Trading Schemes), or voluntary schemes. Offsets are discrete greenhouse gas (GHG) reductions used to compensate for (i.e., offset) GHG emissions elsewhere, for example to meet a voluntary or mandatory GHG target or cap. Offsets are calculated relative to a baseline that represents a hypothetical scenario for what emissions would have been in the absence of the mitigation project that generates the offsets. Examples are carbon offsetting by the Clean Development Mechanism, carbon credits, and other system-external offsets.

Offsets of GHG emissions shall not be included in the inventory nor in the impact assessment of a *Plastics LCA* study, and shall also not be incorporated in the Climate Change indicator result calculated in the study. Offsets may be reported separately as additional environmental information.

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<sup>126</sup> Including landfilling, incineration, composting/digestion followed by on-land application, and in-situ (bio)-degradation.

## 4.5 Handling multi-functional processes

If a process or facility provides more than one function, i.e. it delivers several goods and/or services ("co-products"), it is "multifunctional". In these situations, all inputs and emissions linked to the process shall be partitioned between the product of interest and the other co-products in a principled manner. Systems involving multi-functionality of processes shall be modelled in accordance with the decision hierarchy specified below. However, for husbandry activities at farm and slaughterhouse/rendering activities, the instructions to be used for allocation shall be those specified in Section 4.5.1 of the suggestions for updating the PEF method (Zampori and Pant, 2019). Moreover, specific allocation requirements provided in other sections of this method always prevail over the ones reported in this section (e.g. Section 4.4.7 on transport and logistics, or Section 4.4.15 on greenhouse gas emissions and removals).

### Decision hierarchy

#### 1. Subdivision or system expansion

As per ISO 14044, wherever possible, subdivision or system expansion should be used to avoid allocation. Subdivision refers to disaggregating multifunctional processes or facilities to isolate the input flows directly associated with each process or facility output (i.e. with each product or function). System expansion refers to expanding the analysed system by including additional functions related to the co-products. It shall be investigated first whether the analysed process can be subdivided or expanded. Where subdivision is possible, inventory data should be collected only for those unit processes directly attributable to the goods/services of concern. Alternatively, if the studied system may be expanded, the additional functions shall be included in the analysis with results communicated for the expanded system as a whole rather than on an individual co-product level. However, this alternative may not be compatible with the scope of the study (e.g. if only the main product and its intended function need to be investigated or are relevant for the assessment).

#### 2. Allocation based on a relevant underlying physical relationship

Where it is not possible to apply subdivision or system expansion, allocation should be applied, i.e. the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects relevant underlying (causal) physical relationships between them (ISO 14044:2006).

Allocation based on a relevant underlying physical relationship refers to partitioning the input and output flows of a multi-functional process or facility in accordance with a relevant, quantifiable physical relationship between the process inputs and co-product outputs (for example, a physical property of the inputs and outputs that is relevant to the function provided by the co-product of interest, such as the mass or the energy content). Allocation based on a physical relationship may be modelled in one of the following alternative ways:

- (i) Using *direct substitution*, if it is possible to identify a product that is directly substituted, and the substitution effect is robust.

To demonstrate whether the direct substitution effect is robust, the user of this method shall prove that: (1) there is a direct, empirically demonstrable substitution effect, AND (2) it is possible to model the substituted product and to subtract the respective life cycle inventory in a directly representative manner. If both conditions are fulfilled, the direct substitution effect may be modelled.

Or

- (ii) Allocating input/output flows based on a relevant underlying physical relationship that relates the inputs and outputs to the products or functions provided by the system.

To apply allocation, the user of this method shall demonstrate that it is possible to define a relevant physical relationship by which to allocate the flows attributable to the provision of the relevant product or function of the product system. If this condition is fulfilled, allocation based on the identified physical relationship may be applied.

### 3. Allocation based on some other relationship

Allocation based on some other relationship may be necessary. For example, economic allocation, which refers to allocating inputs and outputs associated with multi-functional processes to the co-product outputs in proportion to their relative market values. The market price of the co-products or co-functions should refer to the specific condition and point in time at which the co-products are produced. Allocation based on economic value shall only be applied when options (1) and (2) are not possible, along the requirements described above. In any case, a clear justification for having discarded options (1) and (2) and for having selected a certain allocation rule in step (3) shall be provided, to ensure the physical representativeness of the LCA results, as far as possible.

Allocation based on some other relationship may be approached in one of the following alternative ways:

- (i) IF it is possible to identify an *indirect substitution* effect<sup>127</sup> AND to model the substituted product and subtract the respective life cycle inventory in a reasonably representative manner, the indirect substitution effect may be modelled.

Or

- (ii) IF it is possible to allocate the input/output flows between the products and functions based on some other relationship (e.g. the relative economic value of the co-products), products and function may be allocated based on the identified relationship.

To summarise, the following multi-functionality decision hierarchy shall be applied for resolving multi-functionality situations: (1) subdivision or system expansion; (2) allocation based on a relevant underlying physical relationship (including direct substitution); and (3) allocation based on some other relationship (including indirect substitution).

All choices made to address multi-functionality, including the chosen solution and the corresponding assumptions and parameters (e.g. allocation factors), shall be reported and justified with respect to the overarching goal of ensuring physically representative, environmentally relevant results.

Dealing with multi-functionality of products is particularly challenging when recycling or energy recovery of one (or more) of these products is involved, as the systems tend to get rather complex. The Circular Footprint Formula (CFF; see Section 4.4.10.2) provides the approach that shall be applied to address multi-functionality in recycling and energy recovery situations, i.e. to estimate the overall emissions associated to a certain process involving recycling and/or energy recovery. This approach also applies to waste flows generated within the system boundary along the product life cycle.

## 4.6 Data collection requirements

This section addresses the data sources and datasets to be used to compile the life cycle inventory of processes and activities included in the system boundary, the corresponding data collection procedures, and how to apply any cut-off.

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<sup>127</sup> Indirect substitution occurs when a product is substituted, but this is not known exactly.

### **4.6.1 Company-specific data**

This section describes the collection of company-specific Life Cycle Inventory data, which are data directly measured or collected at a specific facility or set of facilities, and representative of one or more activities or processes in the system boundary. Section 4.6.1.1 specifies the processes and parameters for which company-specific data shall be collected/calculated and used in the modelling of the Life Cycle Inventory.

The data shall include all known inputs and outputs for the processes. Inputs are (for example) energy, treated/purified water, materials, etc., but also natural resources (as elementary flows), such as land use, groundwater etc. Outputs are the products, co-products, waste generated, and emissions. Emissions occur into any one of the three main compartments (or sub-compartments of these), i.e. air, water and soil.

Company-specific data on material, energy and resource inputs, as well as on waste generation and emission may be collected, measured or calculated using company-specific activity data<sup>128</sup> and appropriate (emission) factors. For instance, emissions from fuel use at a specific facility may be calculated based on the amount of fuel consumed (litre) and emission factors for combustion in a boiler or vehicle). Emission factors may be derived from secondary data that are equally subject to data quality requirements (Sections 4.6.2 and 4.7).

Generally, the most representative sources of data for specific processes are measurements directly performed on the process, or obtained from operators via interviews or questionnaires. However, the measurement frequency may often make measured data less representative than data obtained in other ways, such as via calculations, modelling, from purchase information etc. Collected measured data may need scaling, aggregation or other forms of mathematical treatment to bring them in line with the reference flow of the process and the functional unit of the study.

Typical specific sources of company-specific data are:

- Process- or plant-level consumption data;
- Bills and stock/inventory changes of consumables;
- Emission measurements and/or related calculations (amounts and concentrations of emissions from flue gas and wastewater);
- Composition of products and waste;
- Procurement and sale department(s)/unit(s).

For complex products, the Bill of Material (BoM)<sup>129</sup> is also a valuable source of information. It is constituted of two parts: the list of materials/ingredients (best broken down per assembly, subassembly and/or part or component), and the quantity used for each of them.

#### **4.6.1.1 Requirements**

The activity data of the BoM shall be specific to the product in scope and modelled with company-specific data. For companies producing more than one product the activity data used (including the BoM) shall be specific to the product in scope of the study.

The modelling of the manufacturing processes shall be based on company-specific data (e.g. energy needed for the assembly of the materials/ components of the product in scope). For companies producing more than one product the activity data used (including the BoM) shall be specific to the product in scope of the study.

All company-specific data shall be modelled in company-specific datasets.

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<sup>128</sup> Activity data are data that are specific to the process being considered, as opposed to secondary data.

<sup>129</sup> In some sectors it is equivalent to the bill of components.

All new (company-specific) datasets created when conducting a LCA study conforming to this method shall be EF compliant (see the information available at <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml> for further details on how to develop EF compliant datasets).

#### 4.6.2 Secondary data

Secondary data refers to data that are not based on direct measurements or calculation of the respective processes in the system boundary. Secondary data can be either sector-specific, i.e. specific to the sector being considered for the LCA study, or multi-sector. Examples of secondary data include:

- Industry-average life-cycle data from Life Cycle Inventory databases, industry association reports, government statistics, etc.;
- Data from literature or scientific papers.

When available, sector-specific secondary data shall be used instead of multi-sector secondary data. All secondary data shall be modelled in secondary datasets that shall fulfil the data quality requirements specified in Section 4.7.4. The sources of the data used shall be clearly documented and reported in the LCA report.

#### 4.6.3 Which datasets to use

For processes where no company-specific data are used, *Plastics LCA* studies shall use secondary datasets that are EF compliant<sup>130</sup>, when available. In case an EF compliant secondary dataset does not exist, the selection of the datasets to be used shall be done according to the rules provided below in hierarchical order, i.e. data from a lower level in the hierarchy may only be used if no data set is available on the higher level:

- Use an EF-compliant dataset that can be considered a good proxy for the relevant process or activity. The use of proxy datasets shall be reported in the "Limitations" section of the LCA report. Proxy datasets should be sufficiently close to the actually needed dataset and the data quality requirements specified in Section 4.7.4 apply.
- Use an ILCD-Entry Level (EL) compliant proxy dataset<sup>131</sup> (representative of the relevant process/activity or, in its absence, a good proxy for this). This information shall be reported in the "Limitations" section of the LCA report as data gap. A maximum of 10% of the total environmental impact may be derived from ILCD-EL compliant datasets (calculated cumulatively from lowest to largest contribution to the total environmental profile).
- If no EF-compliant or ILCD-EL compliant proxy is available, then the process shall be excluded from the model. This shall be clearly stated in the "Limitations" section of the LCA report as a data gap.

Both types of gaps shall be explicitly considered in the quantitative/semi-quantitative interpretation of results, and the gap and interpretation shall be validated by the verifier.

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<sup>130</sup> EF compliant datasets are life cycle inventory datasets complying with the modelling rules specified in the PEF method, and with the respective requirements in terms of nomenclature, format and documentation to be used in a dataset. EF compliant datasets provided by the European Commission are available on the nodes listed at <https://eplca.jrc.ec.europa.eu/LCDN/contactListEF.xhtml>, and can be used for free to develop a LCA study under a specific PEFCR (Product Environmental Footprint Category Rule).

<sup>131</sup> In case an ILCD-EL compliant proxy is used, the nomenclature used for the elementary flows shall be aligned with the most recent version of the EF reference package available on the EF developer's page at the following link: <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>. Details to fulfil this aspect are available at <http://eplca.jrc.ec.europa.eu/uploads/MANPROJ-PR-ILCD-Handbook-Nomenclature-and-other-conventions-first-edition-ISBN-fin-v1.0-E.pdf>. Moreover, the EF reference package used for the ILCD-EL proxy dataset shall be the same as the one of EF-compliant datasets used in the LCA study.

#### 4.6.4 Cut-off

Any cut-off shall be avoided, but if not avoidable it is permitted under the following rules:

- Processes (and elementary flows) may be excluded from the LCI model up to a total combined contribution of maximum 3.0% to total mass and energy flows and total environmental impact in all impact categories<sup>132</sup>. This means that processes that in total account for less than 3.0% of the total material and energy flows and potential environmental impact in each impact category may be excluded from a LCA study, starting from the process with the lowest contribution.
- The 3.0% cut-off shall be considered additional to the cut-off already included in the background datasets.
- The cut-off rule is valid for both intermediate and final products.
- The processes subject to cut-off shall be made explicit and justified in the LCA report, in particular with reference to the environmental significance of the cut-off applied.
- A screening study is recommended to identify processes that may be subject to cut-off.

### 4.7 Data quality assessment and quality requirements

This section describes how the data quality of (EF compliant) LCI datasets developed or used in a LCA study shall be assessed, as well as the data quality requirements for these datasets. Data quality requirements are established according to the “materiality” principle, which aims at “focusing on where it really matters”. This means that most relevant lifecycle processes, leading the environmental profile of a product, shall be modelled by using data with higher quality compared to less relevant processes (regardless of where these processes take place in the life cycle of the product).

#### 4.7.1 Data quality requirements and criteria

The data quality requirements and criteria that shall be applied to assess the quality of (EF compliant) LCI datasets are presented in Table 17. They include the following requirements, criteria and aspects:

- Two minimum requirements: (i) *completeness*, and (ii) *methodological appropriateness and consistency* (i.e. full compliance with the PEF method<sup>133</sup>; this can always be assumed for existing EF compliant datasets).
- Once the processes and products that represent the system analysed are chosen, and the LCI of these processes and products are compiled, the completeness criterion evaluates to what degree the LCI covers all the emissions and resources of the processes and products that are required to calculate all the default EF impact categories (consistently adopted in this method). The completeness criterion is a pre-requisite for EF compliant datasets and thus shall not be rated. Full compliance with the PEF method is required for EF compliant datasets, therefore the criterion methodological appropriateness and consistency is also a pre-requisite and shall not be rated.
- Four quality criteria: *technological, geographical, time-related representativeness, and precision*. The representativeness (technological, geographical and time-related) characterises to what degree the processes and products selected are depicting the system analysed, while the precision indicates the way the data is derived and related level of uncertainty. These criteria shall be subject to a scoring procedure.

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<sup>132</sup> During the EF pilot phase, a threshold for cut-off at 1% was applied, generally leading to a high effort for data collection but very limited gains in completeness and accuracy. Therefore, the threshold was increased to 3%, and adopted as a convention in the PEF method (and consistently in this document). In LCA practice, a slightly higher (5%) cut-off is often applied.

<sup>133</sup> And hence also to the present *Plastics LCA* method, which conforms to PEF.

- Three quality aspects: documentation, nomenclature and review. These criteria are not included within the semi-quantitative assessment of the data quality.

**Table 17.** Data quality requirements, criteria and aspects for company-specific and secondary (EF compliant) LCI datasets.

<b>Minimum requirements</b>	<ul style="list-style-type: none"> <li>• Completeness</li> <li>• Methodological appropriateness and consistency <sup>(1)</sup></li> </ul>
<b>Data quality criteria (to be scored)</b>	<ul style="list-style-type: none"> <li>• Technological representativeness (TeR) <sup>(2)</sup></li> <li>• Geographical representativeness (GeR) <sup>(3)</sup></li> <li>• Time-related representativeness (TiR) <sup>(4)</sup></li> <li>• Precision (P)</li> </ul>
<b>Documentation <sup>(5)</sup></b>	<ul style="list-style-type: none"> <li>• Compliant with the ILCD format</li> </ul>
<b>Nomenclature</b>	<ul style="list-style-type: none"> <li>• Compliant with the ILCD nomenclature structure (use of EF reference elementary flows for IT compatible inventories; see detailed requirements at Section 4.3)</li> </ul>
<b>Review <sup>(5)</sup></b>	<ul style="list-style-type: none"> <li>• Review by "Qualified reviewer"</li> <li>• Separate review report</li> </ul>

<sup>(1)</sup> The term "methodological appropriateness and consistency" used throughout this method is equivalent to "consistency" used in ISO 14044.

<sup>(2)</sup> The term "technological representativeness" used throughout this method is equivalent to "technological coverage" used in ISO 14044.

<sup>(3)</sup> The term "geographical representativeness" used throughout this method is equivalent to "geographical coverage" used in ISO 14044.

<sup>(4)</sup> The term "time-related representativeness" used throughout this method is equivalent to "time-related coverage" used in ISO 14044.

<sup>(5)</sup> Detailed requirements regarding documentation and review are provided at <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>.

Each data quality criterion to be scored (TeR, GeR, TiR and P) is rated according to the five levels listed in Table 18.

**Table 18.** Data Quality Rating (DQR) and data quality levels of each data quality criterion.

<b>Data Quality Rating of Data Quality Criteria (TeR, GeR, TiR, P)</b>	<b>Data Quality Level</b>
1	Excellent
2	Very Good
3	Good
4	Fair
5	Poor

### 4.7.2 Semi-quantitative assessment of data quality: the DQR formula

Users of this method shall calculate and report the Data Quality Rating (DQR) of each new EF compliant (LCI) dataset and of the total LCA study. The DQR shall be calculated through a semi-quantitative procedure, based on the four data quality criteria reported in Table 17, and the DQR formula reported below (Equation 9):

$$DQR = \frac{TeR + GeR + TiR + P}{4} \quad \text{[Equation 9]}$$

where TeR is the Technological-Representativeness, GeR is the Geographical-Representativeness, TiR is the Time-Representativeness, and P is Precision.

The application of the DQR formula requires prior calculation of the DQR value (and of the corresponding Data Quality Level) associated with each of the four data quality criteria (TeR, GeR, TiR and P), based on specific scoring criteria. Scoring criteria are context-specific and differs between company-specific datasets and secondary datasets, as described in Section 4.7.3 and 4.7.4, respectively.

Five overall data quality levels (from "Excellent" to "Poor") can be achieved according to the Data Quality Rating (DQR) calculated through the DQR formula. They are summarised in Table 19.

**Table 19.** Overall data quality level of a (EF compliant) LCI dataset according to the achieved data quality rating.

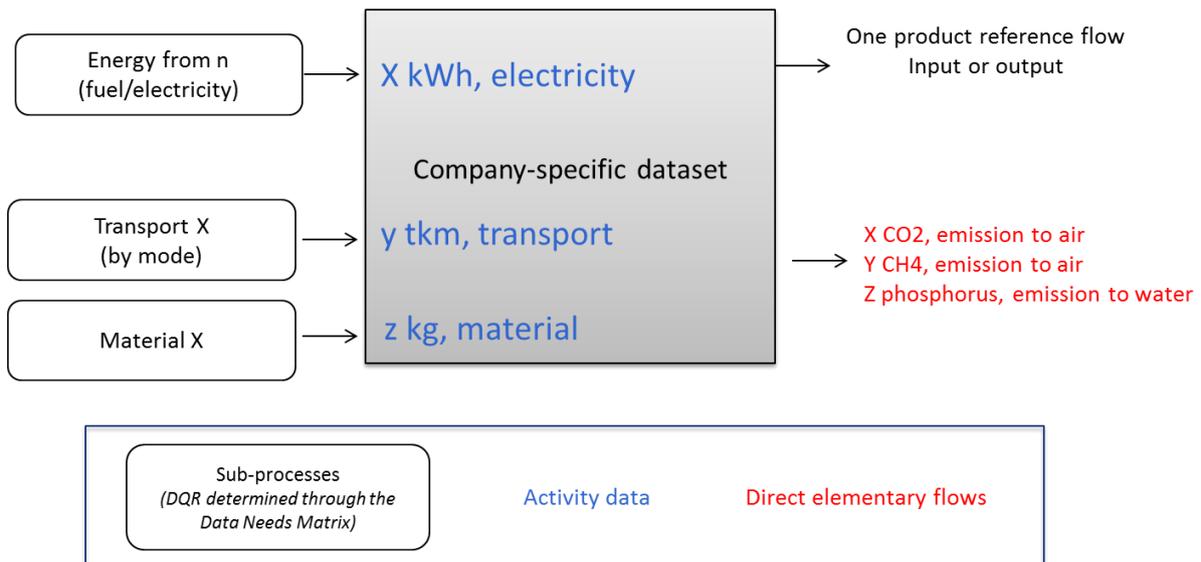
Overall data quality rating (DQR)	Overall data quality level
$DQR \leq 1.5$	"Excellent quality"
$1.5 < DQR \leq 2.0$	"Very good quality"
$2.0 < DQR \leq 3.0$	"Good quality"
$3 < DQR \leq 4.0$	"Fair quality"
$DQR > 4$	"Poor quality"

The DQR formula is applicable to:

1. Company-specific datasets: section 4.7.3 describes the procedure to calculate the DQR of company-specific datasets;
2. Secondary datasets: when using a secondary EF compliant dataset in a LCA study (procedure described in section 4.7.4);
3. The LCA study (procedure described in section 4.7.6).

### 4.7.3 Data quality assessment (DQR) of company-specific datasets

When creating a company-specific dataset, the data quality of i) the company-specific activity data and ii) the company-specific direct elementary flows (i.e. emission data or natural resource input data) shall be assessed separately. The DQR of the sub-processes linked to the activity data (see Figure 14) are evaluated through the requirements provided in the Data Needs Matrix (Section 4.7.5).



**Figure 14.** Graphical representation of a company-specific dataset. A company-specific dataset is a partially disaggregated dataset: the DQR of the activity data and direct elementary flows shall be assessed. The DQR of the sub-processes shall be assessed through the Data Needs Matrix.

The DQR of the newly developed dataset shall be calculated as follow:

1. Select the most relevant activity data and direct elementary flows. Most relevant activity data are those linked to sub-processes (i.e. secondary datasets) that account for at least 80% of the total (weighted) environmental impact of the company-specific dataset, when ranking them from the most contributing to the least contributing one. Most relevant direct elementary flows are defined as those direct elementary flows contributing cumulatively with at least 80% to the total (weighted) impact of the direct elementary flows.
2. Calculate the data quality rate (DQR) value of the data quality (DQR) criteria TeR, TiR, GeR and P for each most relevant activity data (sub-process) and each most relevant direct elementary flow. The value of each DQR criteria shall be assigned based on the scoring criteria reported in Table 20.
  - (a) Each most relevant direct elementary flow consists of the amount and elementary flow naming (e.g. 40 g CO<sub>2</sub>). For each most relevant elementary flow, the practitioner shall evaluate the 4 DQR criteria named Te<sub>R-EF</sub>, Ti<sub>R-EF</sub>, Ge<sub>R-EF</sub>, P<sub>EF</sub> (where EF stands for elementary flow). For example, the practitioner shall evaluate the timing of the flow measured, for which technology the flow was measured and in which geographical area.
  - (b) For each most relevant activity data (sub-process), the 4 DQR criteria named Ti<sub>R-AD</sub>, P<sub>AD</sub>, Ge<sub>R-AD</sub>, Te<sub>R-AD</sub> shall be evaluated (where AD stands for activity data).
  - (c) Considering that both activity data and direct elementary flows shall be company-specific, the score of P cannot be higher than 3, while the score for TiR, TeR and GeR cannot be higher than 2 (the DQR score shall hence be ≤1.5).
3. Calculate the contribution of each most-relevant activity data (sub-process) and direct elementary flow to the total (weighted) environmental impact of all most-relevant activity data (sub-processes) and direct elementary flows of the dataset, in %. For example, if the newly developed dataset has only two most relevant activity data (sub-processes), contributing in total to 80% of the total weighted environmental impact of the dataset, and:

- (a) Activity data (sub-process) 1 carries 30% of the total dataset environmental impact. The contribution of this sub-process to the total of 80% is 37.5% (i.e.  $30/0.8$ ).
  - (b) Activity data (sub-process) 2 carries 50% of the total dataset environmental impact. The contribution of this process to the total of 80% is 62.5% (i.e.  $50/0.8$ ).
  - (c) The calculated values (37.5% and 62.5%) are the weight to be used in the following point 4.
4. Calculate the TeR, TiR, GeR and P criteria of the newly developed dataset as the weighted average of each criteria of the most relevant activity data (sub-processes) and direct elementary flows. The weight is the relative contribution (in %) of each most relevant activity data (sub-process) and direct elementary flow calculated in step 3.
  5. Calculate the total DQR of the newly developed dataset using Equation 10, where  $\overline{Te_R}, \overline{Ge_R}, \overline{Ti_R}, \overline{P}$  are the weighted average calculated as specified in point (4).

$$DQR = \frac{\overline{Te_R} + \overline{Ge_R} + \overline{Ti_R} + \overline{P}}{4} \quad \text{[Equation 10]}$$

**Table 20.** Scoring criteria to assign the values to DQR criteria when using company-specific data. No criteria shall be modified.

Rating	P <sub>EF</sub> and P <sub>AD</sub>	Ti <sub>R-EF</sub> and Ti <sub>R-AD</sub>	Te <sub>R-EF</sub> and Te <sub>R-AD</sub>	Ge <sub>R-EF</sub> and Ge <sub>R-AD</sub>
1	Measured/calculated <u>and</u> externally verified	The data refers to the most recent annual administration period with respect to the EF report publication date	The elementary flows and the activity data exactly the technology of the newly developed dataset	The activity data and elementary flows reflects the exact geography where the process modelled in the newly created dataset takes place
2	Measured/calculated and internally verified, plausibility checked by reviewer	The data refers to maximum 2 annual administration periods with respect to the EF report publication date	The elementary flows and the activity data is a proxy of the technology of the newly developed dataset	The activity data and elementary flows) partly reflects the geography where the process modelled in the newly created dataset takes place
3	Measured/calculated and literature not plausibility checked by reviewer OR Qualified estimate based on calculations plausibility checked by reviewer	The data refers to maximum three annual administration periods with respect to the EF report publication date	Not applicable	Not applicable
4-5	Not applicable	Not applicable	Not applicable	Not applicable

**P<sub>EF</sub>**: Precision for elementary flows; **P<sub>AD</sub>**: Precision for activity data; **Ti<sub>R-EF</sub>**: Time Representativeness for elementary flows; **Ti<sub>R-AD</sub>**: Time representativeness for activity data; **Te<sub>R-EF</sub>**: Technology representativeness for elementary flows; **Te<sub>R-AD</sub>**: Technology representativeness for activity data; **Ge<sub>R-EF</sub>**: Geographical representativeness for elementary flows; **Ge<sub>R-AD</sub>**: Geographical representativeness for activity data.

#### 4.7.4 Data quality assessment (DQR) of secondary datasets

This section describes the procedure to calculate the DQR of secondary datasets used in a *Plastics LCA* study. This means that the DQR of the (EF compliant) secondary dataset (calculated by the data provider) shall be re-calculated, when they are used in the modelling of most relevant processes (see Section 4.7.5), to allow the user of this method to assess the value of context-specific data quality (DQR) criteria (i.e. TeR, TiR and GeR for most relevant processes modelled based on secondary datasets). The TeR, TiR and GeR criteria shall be re-evaluated based on the scoring criteria reported in Table 21. It is not allowed to modify any criteria. The total DQR of the dataset shall be recalculated using the DQR formula (Equation 10).

**Table 21.** Scoring criteria to assign the values to DQR criteria when using secondary datasets. No criteria shall be modified.

Rating	TiR	TeR	GeR
1	The EF report publication date happens within the time validity of the dataset	The technology used in the EF study is exactly the same as the one in scope of the dataset	The process modelled in the EF study takes place in the country the dataset is valid for
2	The EF report publication date happens not later than 2 years beyond the time validity of the dataset	The technologies used in the EF study is included in the mix of technologies in scope of the dataset	The process modelled in the EF study takes place in the geographical region (e.g. Europe) the dataset is valid for
3	The EF report publication date happens not later than 4 years beyond the time validity of the dataset	The technologies used in the EF study are only partly included in the scope of the dataset	The process modelled in the EF study takes place in one of the geographical regions the dataset is valid for
4	The EF report publication date happens not later than 6 years beyond the time validity of the dataset	The technologies used in the EF study are similar to those included in the scope of the dataset	The process modelled in the EF study takes place in a country that is not included in the geographical region(s) the dataset is valid for, but sufficient similarities are estimated based on expert judgement.
5	The EF report publication date happens later than 6 years after the time validity of the dataset, or the time validity is not specified	The technologies used in the EF study are different from those included in the scope of the dataset	The process modelled in the EF study takes place in a different country than the one the dataset is valid for

**TiR:** Time representativeness; **TeR:** Technology representativeness; **GeR:** Geographic representativeness.

#### **4.7.5 Data quality requirements: the Data Needs Matrix (DNM)**

The Data Needs Matrix shall be used to evaluate all processes required to model the product in scope based on their data requirements (see Table 22). It indicates for which processes company-specific data or secondary data shall or may be used, depending on the level of influence the company has on the process. The following three cases are found in the DNM and explained below:

1. **Situation 1:** the process is run by the company performing the LCA study.
2. **Situation 2:** the process is not run by the company performing the LCA study, but the company has access to (company)-specific information.
3. **Situation 3:** the process is not run by the company performing the LCA study and this company does not have access to (company)-specific information.

The user of this method shall:

1. Determine the level of influence (Situation 1, 2 or 3) the company has for each process in its supply chain. This decision determines which of the options in Table 22 is pertinent for each process;
2. Provide a table in the LCA report listing all processes and their situation according to the DNM;
3. Follow the data requirements indicated in Table 22;
4. Calculate / re-evaluate the DQR values (for each criterion + total) for the datasets of most relevant processes and the new ones created, as indicated in Sections 4.7.5.1 – 4.7.5.3.

**Table 22.** Data Needs Matrix (DNM) – Requirements for a company performing a *Plastics LCA* study. The options indicated for each situation are not listed in hierarchical order.

		<b>Data requirements</b>
<b>Situation 1:</b> process run by the company	<b>Option 1</b>	Provide company-specific data (both activity data and direct emissions) and create a company-specific dataset ( $DQR \leq 1.5$ ). Calculate DQR of the dataset following the rules at Section 4.7.3.
<b>Situation 2:</b> process <u>not</u> run by the company but with access to company-specific information	<b>Option 1</b>	Provide company-specific data and create a company-specific dataset ( $DQR \leq 1.5$ ). Calculate DQR of the dataset following the rules at Section 4.7.3.
	<b>Option 2</b>	Use an EF-compliant secondary dataset and apply company-specific activity data for transport (distance), and substitute the sub-processes used for electricity mix and transport with supply-chain specific EF compliant datasets ( $DQR \leq 3.0$ ). Recalculate DQR of the dataset used (see Section 4.7.5.2).
<b>Situation 3:</b> process <u>not</u> run by the company and without access to company-specific information	<b>Option 1</b>	Use an EF-compliant secondary data set in aggregated form ( $DQR \leq 3.0$ ). Recalculate DQR of the dataset if the process is most relevant (see Section 4.7.5.3)

#### **4.7.5.1 DNM, situation 1**

For all processes run by the company and where the company performing the LCA study uses company-specific data, the DQR of the newly developed (EF compliant) LCI dataset shall be evaluated as described in Section 4.7.3.

#### **4.7.5.2 DNM, situation 2**

When a process is in situation 2 (i.e. the company performing the LCA study is not running the process but has access to company-specific data) there are two possible options:

- The user of this method has access to extensive supplier-specific information and wants to create a new EF-compliant dataset (Option 1);
- The company has some supplier-specific information and wants to make some minimum changes (Option 2);

#### **Situation 2/Option 1**

For all processes not run by the company and where the company performing the LCA study uses company-specific data, the DQR of the newly developed EF compliant dataset shall be evaluated as described in Section 4.7.3.

## **Situation 2/Option 2**

A disaggregated secondary EF compliant dataset is used for processes in Situation 2/Option 2. The company performing the LCA study shall:

- Use company-specific activity data for transport;
- Substitute the sub-processes for the electricity mix and transport used in the disaggregated secondary EF compliant dataset with supply chain specific EF compliant datasets.

Supply-chain specific  $R_1$  values may be used. The user of this method shall recalculate the DQR criteria for the processes in Situation 2, Option 2. It shall make the DQR context-specific by re-evaluating TeR and TiR using the criteria provided in Table 21. The criterion GeR shall be lowered by 30% and the criterion P shall keep the original value.

### **4.7.5.3 DNM, situation 3**

If a process is in situation 3 (i.e. the company performing the LCA study is not running the process and this company does not have access to company-specific data), the company performing the LCA study shall use EF compliant secondary datasets.

If the process is a most relevant one, following the procedure described in Section 6.2.3, the user of this method shall make the DQR criteria context-specific by re-evaluating TeR, TiR and GeR using Table 21. The parameter P shall keep the original value.

For the non-most relevant processes, following the procedure described in Section 6.2.3, the company performing the LCA study shall take the DQR values from the original dataset.

### **4.7.6 Data Quality Assessment (DQR) of a LCA study**

The DQR of the LCA study (i.e. of the overall EF compliant dataset for the product in scope) shall be calculated and reported in the LCA study report.

To calculate the DQR of the LCA study, the user of this method shall first calculate separately the TeR, TiR, GeR and P of the study as the weighted average of the values of TeR, TiR and GeR related to all most relevant processes<sup>134</sup>. Weighting factors to be used in calculation shall be based on the relative environmental contribution (in %) of the specific most relevant process to the total weighted impact score (single score) from all most relevant processes. The DQR calculation rules reported in section 4.7.3 (Equation 10) shall then be followed.

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<sup>134</sup> Most relevant processes are those that collectively contribute with at least **80%** to any of the considered impact categories.

## 5 Life Cycle Impact Assessment

Once the Life Cycle Inventory has been compiled, the Life Cycle Impact Assessment (LCIA)<sup>135</sup> shall be undertaken to calculate the environmental performance of the product, using all the default impact categories and models (entirely conforming to those applied in the PEF context; Section 3.2.4). Life Cycle Impact Assessment includes four steps: classification, characterisation, normalisation and weighting.

Results of a *Plastics LCA* study shall be calculated and reported in the LCA report as characterised, normalised and weighted results for each impact category, and as a single overall score to be calculated based on the normalisation and weighting factors specified in Sections 5.2.1 and 5.2.2. Results shall be reported for (i) the total life cycle, and (ii) the total life cycle excluding the Use stage. For intermediate products, the Use stage and End of Life are excluded.

### 5.1 Classification and Characterisation

Life Cycle Impact Assessment shall include a classification and characterisation of the Life Cycle Inventory (LCI) flows (i.e. elementary flows occurring throughout the whole product life cycle determined in the LCI).

The steps of classification and characterisation are usually pre-implemented in the vast majority of software and tools that may be used to conduct a LCA study, and do not need any specific action or calculation by the user. However, the logic behind these steps is briefly explained in the following sections, to allow the user of this method to better understand the interpretation of the corresponding results.

#### 5.1.1 Classification

Classification requires assigning the material/energy inputs and outputs compiled in the Life Cycle Inventory (LCI) to the relevant LCA impact category. For example, during the classification phase, all inputs/outputs that result in greenhouse gas emissions are assigned to the Climate Change category. Similarly, those that result in emissions of ozone-depleting substances are classified accordingly to the Ozone Depletion category. In some cases, an input/output may contribute to more than one impact category (for example, chlorofluorocarbons (CFCs) contribute to both Climate Change and Ozone Depletion).

All inputs and outputs inventoried during the compilation of the LCI shall be assigned to the impact categories to which they contribute ("classification"), using the classification data available at <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>.

For a proper classification of the Life Cycle Inventory, LCI data (i.e. elementary flows) should be expressed in terms of specific (constituent) substances for which characterisation factors are available (see Section 5.1.2), as far as possible<sup>136</sup>. This means that elementary flows used in the LCI should be in line with the default list of elementary flows reported in the most recent version of the EF reference package available at the time of the study, and accessible at <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>.

<sup>135</sup> The Life Cycle Impact Assessment does not intend to replace other (regulatory) tools that have a different scope and objective such as (Environmental) Risk Assessment ((E)RA), site-specific Environmental Impact Assessment (EIA), or Health and Safety regulations at product level or related to safety at the workplace. Especially, the Life Cycle Impact Assessment has not the objective to predict if, at any specific location and at any specific time, thresholds are exceeded and actual impacts occur. In contrast, it describes the cumulated potential effects (impacts) from existing pressures on the environment occurring throughout the whole product life cycle and the locations where the corresponding activities take place. This is made by aggregating lifecycle emissions and natural resource consumption across time and space. Thus, the Life Cycle Impact Assessment is complementary to other well-proven tools, adding the life cycle perspective.

<sup>136</sup> For example, if in a given impact category the characterisation factors are available for nutrients (e.g. N and P) directly applied to agricultural soil, LCI data related to the amount of e.g. a composite NPK fertiliser applied to the soil shall be disaggregated, inventoried and classified according to its N, P and K fractions. This is also because each constituent element may contribute to different impact categories.

In practice, when conducting a LCA study, much of the Life Cycle Inventory data may be drawn from existing public or commercial LCI databases, where classification has already been implemented. In such cases, it must be assured, for example by the provider, that the classification and linked impact assessment pathways correspond to the requirements of this method.

### **5.1.2 Characterisation**

Characterisation refers to the calculation of the magnitude of the contribution of each classified input and output to the respective impact categories, and aggregation of the contributions within each category. This is carried out by multiplying the values reported in the Life Cycle Inventory (LCI) for each elementary flows by the relevant characterisation factor (CF) for each impact category.

The characterisation factors are substance- or resource-specific. They represent the impact intensity of a substance relative to a common reference substance for an impact category (and the related impact category indicator). For example, in the case of calculating the Climate Change impact indicator, all greenhouse gas emissions inventoried in the LCI are "weighted" in terms of their impact intensity relative to carbon dioxide, which is the reference substance for this category. This enable the aggregation of impact potentials from different substances, and their expression in terms of a single equivalent substance for each impact category (in the case of Climate Change, in terms of CO<sub>2</sub> equivalents). For example, the CF expressed as Global Warming Potential over a 100-year time horizon (GWP<sub>100</sub>) for the flow "methane, fossil" equals 36.75 kg of CO<sub>2</sub>-equivalents (IPCC, 2013) and its contribution to the Climate Change impact is thus 36.75 times higher than the one of "CO<sub>2</sub>, fossil" (having a CF of 1 kg of CO<sub>2</sub>-equivalents).

All classified inputs and outputs in each impact category shall be assigned characterisation factors representing the contribution per unit of input or output to the category, using the provided characterisation factors available online at <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>. Life cycle impact assessment results shall subsequently be calculated for each impact category by multiplying the amount of each input and output by its characterisation factor and summing the contributions of all inputs and outputs within each category to obtain a single measure expressed in the appropriate reference unit.

## **5.2 Normalisation and Weighting**

Following the steps of classification and characterisation, the Life Cycle Impact Assessment shall be complemented with normalisation and weighting.

Similarly to classification and characterisation, also the steps of normalisation and weighting are usually pre-implemented in the vast majority of software and tools that may be used to conduct a LCA study, and do not need any specific action or calculation by the user. However, the logic behind these steps is briefly explained in the following sections, to allow the user of this method to better understand the interpretation of the corresponding results.

### **5.2.1 Normalisation of Life Cycle Impact Assessment Results**

Normalisation is the step in which the characterised Life Cycle Impact Assessment results are multiplied by normalisation factors, to calculate and compare the magnitude of their contributions to the impact categories relative to a reference unit (such as the total potential impact caused over one year in a specific region or country, or from an average person in such region or country). As a result, dimensionless, normalised results are obtained. These results reflect the impacts attributable to a product relative to the chosen reference unit, such as per capita for a given year and region. They can be used, for instance, to evaluate the contribution of the product (or of individual life cycle stages or processes) to the impact level associated with the reference unit. For example, LCIA results of the product may be compared to the potential impacts calculated for a given

region (such as EU-27) on a per-person and per-year basis, which are used as a reference unit. Normalised results do not, however, indicate the severity or relevance of the respective impacts.

Normalisation is a required step for LCA studies applying this method. In line with the PEF method, the normalisation factors adopted in this method are expressed per capita, based on global values. The set of normalisation factors that shall be applied is the one reported in the most recent version of the EF reference package available at the time of the study (currently 3.0), and provided at <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>.

Normalised LCA results shall not be aggregated, since this implicitly applies weighting (which shall be performed as described in Section 5.2.2). Normalised results shall be reported in the LCA report alongside characterised results calculated prior to normalisation.

### **5.2.2 Weighting of Life Cycle Impact Assessment Results**

Weighting is a mandatory step in LCA studies conforming to this method, and it supports the interpretation and communication of the results. In this step, normalised results are multiplied by a set of weighting factors (in %) which reflect the perceived relative importance of the considered impact categories. Weighted results of different impact categories may then be compared to assess their relative importance. They may also be aggregated across the different impact categories to obtain a single overall impact score.

To develop weighting factors, value judgements are required as to the relative importance of the considered impact categories. The weighting factors<sup>137</sup> that shall be applied are those provided in the most recent version of the EF reference package available at the time of the study (currently 3.0), and provided at <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>.

Weighted LCA results shall be reported in the LCA report alongside characterised results calculated prior to normalisation and weighting.

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<sup>137</sup> For more information on weighting approaches applied in the PEF context and the calculation of corresponding weighting factors, please refer to the reports developed by the JRC available online at: [http://ec.europa.eu/environment/eussd/smgp/documents/2018\\_JRC\\_Weighting\\_EF.pdf](http://ec.europa.eu/environment/eussd/smgp/documents/2018_JRC_Weighting_EF.pdf)

## 6 Interpretation of LCA results

Interpretation of the results of the LCA<sup>138</sup> study serves two purposes:

- The first is to ensure that the LCA model corresponds to the goals of the study and fulfil its quality requirements. In this sense, results interpretation may inform iterative improvements of the LCA model until all goals and requirements are met;
- The second purpose is to derive robust conclusions and recommendations from the analysis, for example in support of environmental improvements.

To meet these objectives, the result interpretation phase shall include the steps outlined in this section.

### 6.1 Assessment of the robustness of the LCA model

The assessment of the robustness of the LCA model assesses the extent to which methodological choices such as the system boundary, data sources, allocation choices, influence the outcomes of the analysis.

Tools that should be used to assess the robustness of the LCA model include:

- **Completeness checks:** assess the Life Cycle Inventory data to ensure that it is complete relative to the defined goals, scope, system boundary and quality criteria. This includes completeness of process coverage (i.e. all processes at each supply-chain stage considered have been included) and input/output coverage (i.e. all relevant material and/or energy inputs and emissions associated with each process have been included).
- **Sensitivity checks:** assess the extent to which the results are determined by specific methodological choices and case-wise relevant assumptions<sup>139</sup>, and the impact of implementing alternative choices or assumptions where these are identifiable (especially focusing on more influencing choices and/or assumptions that are affected by larger uncertainty). It is useful to structure sensitivity checks for each phase of the LCA study, including goal and scope definition, the Life Cycle Inventory, and the Life Cycle Impact Assessment.
- **Consistency checks:** assess the extent to which assumptions, methods, common background data, and data quality considerations have been applied consistently throughout the LCA study.

Any issues flagged in this evaluation may be used to inform iterative improvements to the LCA study.

### 6.2 Identification of Hotspots: most relevant impact categories, life cycle stages, processes and elementary flows

Once it has been ensured that the LCA model is robust and conforms to all aspects defined in the goal and scope definition phases, the main contributing elements to the LCA results shall be identified. This step may also be referred to as "hotspot" analysis. Contributing elements may be specific life-cycle stages, processes, or individual natural resource inputs or emissions (i.e. elementary flows) associated with a given stage or process in the product life cycle. Single impact categories can also be considered as contributing elements to the total single impact score calculated via weighting. The analysis of main contributing elements provides the necessary basis to identify possible improvement potentials of the environmental performance of the product, associated with specific management interventions.

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<sup>138</sup> The term "life cycle interpretation" is used in ISO 14044 to refer to this stage.

<sup>139</sup> Examples of case-wise relevant assumptions that (may) considerably affect the results and that may be considered for sensitivity analysis include assumptions related to the configuration of the supply-chain, the calculation of the reference flow, transport distances, applied end of life shares (e.g. recycling rates).

The user of this method shall identify and list in the LCA report (along with the respective % contribution) the most relevant:

1. Impact categories,
2. Life cycle stages,
3. Processes, and
4. Elementary flows.

The procedure that shall be followed to identify the most relevant impact categories, life cycle stages, processes and direct elementary flows is described in the following sections (6.2.1-6.2.7).

There is an important operational difference between most relevant impact categories and life cycle stages on one hand, and most relevant processes and elementary flows on the other. In particular, most relevant impact categories and life cycle stages may be mainly relevant in the context of the "communication" of the results of a LCA study. They might also serve the purpose of "warning" an organisation about the areas where they should focus their attention.

The identification of the most relevant processes and elementary flows is more important for engineers and designers to identify actions for improving the overall environmental performance of the product (such as by-passing or changing a process, further optimising a process, apply pollution control technology, etc.). This is particularly relevant for internal studies, to look deeper on how to improve the product environmental performance.

### **6.2.1 Procedure to identify the most relevant impact categories**

The identification of the most relevant impact categories shall be based on the normalised and weighted results. The most relevant impact categories shall be identified as all impact categories that cumulatively contribute to at least **80%** of the total environmental impact (single score). This shall start from the largest to the smallest contributions.

At least three relevant impact categories shall be identified as most relevant ones. The user of this method may add more impact categories to the list of the most relevant ones, but none of them shall be deleted.

### **6.2.2 Procedure to identify the most relevant life cycle stages**

Most relevant life cycle stages shall be identified for each of the most relevant impact categories identified as described in Section 6.2.1. The most relevant life cycle stages are the ones that together contribute to at least **80%** of the total impact in any of the most relevant impact categories identified. This shall start from the largest to the smallest contributions. The user of this method may add more life cycle stages to the list of the most relevant ones, but none of them shall be deleted. As a minimum, the five life cycle stages addressed in Section 4.2 shall be considered (i.e. Raw Material Acquisition and Pre-processing, Manufacturing, Distribution stage, Use stage, and End of Life).

If the Use stage accounts for more than 50% of the total impact, then the procedure shall be re-run by excluding the Use stage. In this case, the list of most relevant life cycle stages shall include the ones selected through this second procedure, AND the Use stage.

### **6.2.3 Procedure to identify the most relevant processes**

Each most relevant impact category shall be further investigated to identify the most relevant processes used to model the product life cycle. The most relevant processes are those that collectively contribute to at least **80%** of the total impact in any of the most

relevant impact categories identified as described in Section 6.2.1. Identical processes<sup>140</sup> (or similar processes providing the same function) taking place in different life cycle stages (e.g. transportation, electricity use) shall be accounted for separately. Identical processes taking place within the same life cycle stage shall be accounted for together. The list of most relevant processes shall be reported in the LCA study report together with the respective life cycle stage (or multiple life cycle stages if relevant) and the respective % contribution. The list shall be reported separately for each most relevant impact category. The user of this method may add more processes to the list of the most relevant ones, but none of them shall be deleted. The identification of the most relevant processes shall be done according to Table 23, depending on the contribution of the Use stage to the total impact in a specific impact category.

**Table 23.** Criteria to select at which life cycle stage level to identify the most relevant processes.

<b>Contribution of the Use stage to the total impact</b>	<b>Most relevant processes identified at the level of:</b>
≥ 50%	Whole life cycle <u>excluding</u> the Use stage, AND Use stage separately
< 50%	Whole life cycle

#### 6.2.4 Procedure to identify the most relevant elementary flows

The most relevant elementary flows are defined as those flows contributing cumulatively at least with **80%** to the total impact of each most relevant process, starting from the most contributing to the less contributing ones. This analysis shall be conducted and reported separately for each most relevant impact category.

Elementary flows belonging to the background system of a most relevant process may dominate the total impact. Therefore, if disaggregated datasets are available, the user of this method should in addition identify the most relevant direct elementary flows for each most relevant process.

Most relevant direct elementary flows are defined as those flows contributing cumulatively at least with **80%** to the total impact of the direct elementary flows of the process, for each most relevant impact category. The analysis shall be limited to the direct emissions of the level-1 disaggregated datasets (see <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml> for a description of level-1 disaggregated datasets). This means that the 80% cumulative contribution shall be calculated against the impact caused by the direct emissions only, and not against the total impact of the process.

The user of this method may add more elementary flows to the list of the most relevant ones, but none of them shall be deleted. The list of most relevant elementary flows (or, if applicable, direct elementary flows) per most relevant process shall be reported in the LCA report.

#### 6.2.5 Dealing with negative numbers

When identifying the percentage impact contribution for any process or elementary flow, it is important that absolute values are used (i.e. the minus sign, if present, is ignored). This permit to identify the relevance of any credits (e.g., from recycling). In case of processes or flows with a negative impact score, the following procedure shall be applied:

<sup>140</sup> Two processes are identical when they have the same UUID.

- Consider the absolute values (i.e. the impacts of processes or flows to have a plus sign, namely a positive score);
- Recalculate the total impact score including the converted negative scores (i.e. as positive scores);
- Set the total impact score to 100%; and
- Evaluate the percentage impact contribution of any life cycle stage, process or flow with respect to this new total.

This procedure does not apply to identify the most relevant life cycle stages.

## 6.2.6 Summary of requirements

Table 24 summarises the requirements to identify most relevant contributions (i.e. most relevant impact categories, life cycle stages, processes and elementary flows).

**Table 24.** Summary of requirements to define most relevant contributions.

Item	At what level shall relevance be identified?	Threshold
<b>Most relevant impact categories</b>	Total single impact score, based on normalised and weighted results	Impact categories cumulatively contributing to at least <b>80%</b> of the total environmental impact of the product (single score)
<b>Most relevant life cycle stages</b>	For each most relevant impact category	All life cycle stages contributing cumulatively to more than <b>80%</b> of the total impact in the specific impact category <sup>(1)</sup>
<b>Most relevant processes</b>	For each most relevant impact category	All processes contributing cumulatively to more than <b>80%</b> of the total impact in the specific impact category, considering all processes included in the product life cycle and absolute values of the respective impacts <sup>(1)</sup>
<b>Most relevant elementary flows</b>	For each most relevant process and most relevant impact category	All elementary flows contributing cumulatively at least to 80% of the total impact of each most relevant process in the specific impact category If disaggregated datasets are available: for each most relevant process, all direct elementary flows contributing cumulatively at least to <b>80%</b> of the total impact caused by direct elementary flow of the process in the specific impact category

<sup>(1)</sup> Different procedures apply, depending on the contribution of the Use stage (see Sections 6.2.2 and 6.2.3).

## 6.2.7 Example

This section provides fictitious examples (not based on any specific LCA study results) of identification of most relevant contributions (impact categories, life cycle stages, processes and elementary flows). The examples are illustrated in Tables 25-28.

### **Most relevant impact categories**

**Table 25.** Contribution of different impact categories based on normalised and weighted LCA results - example.

<b>Impact Category</b>	<b>Contribution to the total impact – single score (%)</b>
Climate Change	21.5
Ozone Depletion	3.0
Human Toxicity - cancer	6.0
Human Toxicity - non-cancer	0.1
Particulate Matter	14.9
Ionizing Radiation	0.5
Photochemical Ozone Formation	2.4
Acidification	1.5
Eutrophication - terrestrial	1.0
Eutrophication - freshwater	1.0
Eutrophication - marine	0.1
Ecotoxicity - freshwater	0.1
Land Use	14.3
Water Use	18.6
Resource Use - minerals and metals	6.7
Resource Use - fossils	8.3
<b>Total of most relevant Impact Categories</b>	<b>84.3</b>

Based on the normalised and weighted results, the most relevant impact categories are: Climate Change, Particulate Matter, Water Use, Land Use, Resource Use (minerals and metals), and Resource Use (fossils), for a total cumulative contribution equal to 84.3% of the total normalised and weighted impact (single score).

**Most relevant life cycle stages**

**Table 26.** Contribution of different life cycle stages to the Climate Change impact category (based on characterised LCA results) - example.

<b>Life cycle stage (LCS)</b>	<b>Contribution (%)</b>
Raw material acquisition and pre-processing	46.3
Manufacturing (of the main product)	21.2
Distribution	16.5
Use stage	5.9
End of Life	10.1
<b>Total of most relevant LCS</b>	<b>88.0</b>

The three life cycle stages in purple are the ones identified as "most relevant" for the Climate Change impact category, as they are contributing to more than 80% of the total normalised and weighted impact in this category. Ranking shall start from the highest contributors.

This procedure shall be repeated for all the identified most relevant impact categories.

### **Most relevant processes**

**Table 27.** Contribution of different processes to the Climate Change impact category (based on characterised LCA results).

<b>Life cycle stage</b>	<b>Unit process</b>	<b>Contribution (%)</b>
Raw material acquisition and pre-processing	Process A	4.9
	Process B	41.4
Manufacturing (of the main product)	Process C	18.4
	Process D	2.8
Distribution	Process E	16.5
Use stage	Process F	5.9
End of Life	Process G	10.1
<b>Total of most relevant processes</b>		<b>86.4</b>

According to the described procedure (Section 6.2.3), the processes B, C, E and G shall be selected as “most relevant”.

This procedure shall be repeated for all the identified most relevant impact categories.

## Dealing with negative numbers and identical processes in different life cycle stages

**Table 28.** Example on how to deal with negative numbers and identical process in different life cycle stages.

Impact Category 1 (Characterised results)

1. Characterised results of a most relevant EF  
Impact Category

	LC stage 1	LC stage 2	LC stage 3	LC stage 4	LC stage 5	total per process	% per process
Process A	18	23				41	44%
Process B			13			13	14%
Process C	17				-9	8	9%
Process D	5			6		11	12%
Process E	4	4	4	4	4	20	22%
Total of LC						93	100%

2. Convert everything to absolute values

	LC stage 1	LC stage 2	LC stage 3	LC stage 4	LC stage 5	total per process	% per process
Process A	18	23				41	38%
Process B			10			10	9%
Process C	17				9	26	24%
Process D	5			6		11	10%
Process E	4	4	4	4	4	20	19%
Total of LC						108	100%

Most relevant processes

3. Calculate the % per process and life cycle stage

	LC stage 1	LC stage 2	LC stage 3	LC stage 4	LC stage 5	total per process (absolute values)	% per process
Process A	17%	21%				41	38%
Process B			9%			10	9%
Process C	16%				8%	26	24%
Process D	5%			6%		11	10%
Process E	4%	4%	4%	4%	4%	20	19%
Total of LC						108	100%

## 6.3 Conclusions, Recommendations and Limitations

The final step of the LCA result interpretation phase is to draw conclusions based on the results of the analysis, answer the questions posed at the outset of the LCA study (goal definition), and advance recommendations appropriate for the intended audience and context, while explicitly taking into account any limitations to the robustness and applicability of the results. The LCA study needs to be seen as complementary to other assessments and instruments such as site-specific environmental impact assessment or chemical risk assessment.

Potential improvement options can and should also be identified, such as using cleaner technology or production techniques, changes in product design, implementation of environmental management systems (e.g. Eco-Management and Audit Scheme (EMAS) or ISO 14001), or other systematic approaches.

Conclusions, recommendations and limitations shall be described in accordance with the defined goals and scope of the LCA study. The conclusions should include a summary of identified supply chain "hotspots" and, where appropriate, indicate the potential improvement in the environmental performance of the product(s) associated with specific management interventions.

## **7 LCA report**

### **7.1 Introduction**

A LCA report complements the LCA study and provides a relevant, comprehensive, consistent, accurate, and transparent summary of the study itself. It reflects the best possible information in such a way as to maximise its usefulness to intended current and future users, while transparently communicating the limitations. Effective LCA reporting requires that several criteria, both procedural (report quality) and substantive (report content), are met. A LCA report template is available in Annex E. The template includes the minimum information to be reported in a LCA report.

A LCA report consists of at least: a summary, the main report, the aggregated EF compliant dataset of the study, and an annex. Confidential and proprietary information may be documented in a fourth element - a complementary confidential report. Review reports are annexed.

#### **7.1.1 Summary**

The summary shall be able to stand alone without compromising the results and conclusions/recommendations (if included). The summary shall fulfil the same criteria about transparency, consistency, etc. as the detailed report. To the extent possible, the summary should be written targeting a non-technical audience.

#### **7.1.2 Main report**

The main report<sup>141</sup> shall, as a minimum, include the following components:

- General information,
- Goal of the study,
- Scope of the study,
- Life Cycle Inventory analysis,
- Life Cycle Impact Assessment results,
- Interpretation of the LCA results.

#### **7.1.3 Aggregated EF compliant dataset**

For each product included in the scope of the LCA study, the user of this method shall make available an aggregated EF compliant dataset to the European Commission. The details related to the use and intellectual property rights related to this data are available at the link: [http://ec.europa.eu/environment/eussd/smgp/pdf/IPR\\_PEFCSR\\_OEFSR.pdf](http://ec.europa.eu/environment/eussd/smgp/pdf/IPR_PEFCSR_OEFSR.pdf).

If the user of this method publishes such an EF compliant dataset, the LCA report of the study from which the dataset has been generated shall also be made public.

#### **7.1.4 Validation statement**

See Section 8.5.3.

#### **7.1.5 Annexes**

The annexes serve to document supporting elements to the main report, which are of a more technical nature (e.g. detailed calculations for data quality assessment, results of sensitivity analysis, assessment of the robustness of the LCA model, bibliographic references).

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<sup>141</sup> The main report, as defined here, is insofar as possible in line with ISO 14044 requirements on reporting for studies that do not contain comparative assertions to be disclosed to the public.

### **7.1.6 Confidential report**

The confidential report is an optional reporting element that shall contain all data (including raw data) and information that are confidential or proprietary and may not be made externally available. The confidential report shall be made available for the verification and validation procedure of the LCA study (see Section 8.4.3).

## 8 Verification and validation of LCA studies, reports and communication vehicles

In case policies implementing this method define specific requirements regarding verification and validation of compliant LCA studies, reports and communication vehicles, the requirements in said policies shall prevail.

### 8.1 Defining the scope of the verification

The verification and validation of the LCA study is mandatory whenever the LCA study, or part of the information therein, is used for any type of external communication (i.e. communication to any interested party other than the commissioner of the study or the user of this method).

**Verification** means the conformity assessment process carried out by a verifier to check whether the LCA study has been carried out in compliance with the most updated version of the *Plastics LCA* method.

**Validation** means the confirmation by the verifier who carried out the verification, that the information and data included in the LCA study, the LCA study report and the communication vehicles are reliable, credible and correct.

The verification and validation shall cover the following three areas:

1. The LCA study (including, but not limited to the data collected, calculated, and estimated and the underlying model);
2. The LCA report;
3. The technical content of the communication vehicles, if applicable.

The verification of the **LCA study** shall ensure that the LCA study is conducted in compliance with the most recent version of the *Plastics LCA* method.

The validation of information in the LCA study shall ensure that:

1. The data and information used for the LCA study are consistent, reliable and traceable;
2. The calculations performed do not include significant<sup>142</sup> mistakes.

The verification and validation of the **LCA report** shall ensure that:

1. The LCA report is complete, consistent, and compliant with the LCA report template provided in the most recent version of the *Plastics LCA* method;
2. The information and data included are consistent, reliable and traceable;
3. The mandatory information and sections are included and appropriately filled in;
4. All the technical information that could be used for communication purposes, independently from the communication vehicle to be used, are included in the report.

**Note:** confidential information shall be subject to validation, while they may be excluded from the LCA report.

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<sup>142</sup> Mistakes are significant if they change the final result by more than 5% for any of the impact categories, or the identified most relevant impact categories, life cycle stages and processes.

The validation of the technical content of the **communication vehicle** content shall ensure that:

1. The technical information and data included are reliable and consistent with the information included in the LCA study and in the LCA report;
2. That the information is compliant with the requirements of the Unfair Commercial Practices Directive<sup>143</sup>;
3. That the communication vehicle fulfils the principles of transparency, availability and accessibility, reliability, completeness, comparability and clarity, as described in the Commission Communication on Building the Single Market for Green Products<sup>144</sup>.

## 8.2 Verification procedure

The verification procedure covers the following steps:

1. The commissioner shall select the verifier or verification team following the rules outlined in Section 8.3.1;
2. The verification shall be performed following the verification process described in Section 8.4;
3. The verifier shall communicate to the commissioner any misstatements, non-conformities and need for clarifications (Section 8.3.2), and draft the validation statement (Section 8.5.2);
4. The commissioner shall respond to the verifier's comments and introduce necessary corrections and changes (if needed) to ensure the final compliance of the LCA study, the LCA report and the technical content of the LCA communication vehicles. If, in the verifier's judgement, the commissioner does not respond appropriately within a reasonable time period, the verifier shall issue a modified validation statement;
5. The final validation statement is provided, considering (if needed) the corrections and changes introduced by the commissioner;
6. Surveillance that the LCA report is available during the validity of the validation statement (as defined in Section 8.5.3).

If a matter comes to the verifier's attention that causes the verifier to believe in the existence of fraud or noncompliance with laws or regulations, the verifier shall communicate this immediately to the commissioner of the study.

## 8.3 Verifier(s)

The verification/validation may be performed by a single verifier or by a verification team. The independent verifier(s) shall be external to the organisation that conducted the LCA study.

In all cases, the independence of the verifiers shall be guaranteed, i.e. they shall fulfil the intentions in the requirements of ISO/IEC 17020:2012 regarding a 3<sup>rd</sup> party verifier, and they shall not have conflicts of interests on concerned products.

In case the LCA study is done based on a PEFCR, verifiers shall not include members of the Technical Secretariat or of the consultants involved in previous parts of the work (i.e. screening studies, supporting studies, PEFCR review, etc.).

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<sup>143</sup> [Directive 2005/29/EC](#) of the European Parliament and of the Council of 11 May 2005 concerning unfair business-to-consumer commercial practices in the internal market and amending Council Directive 84/450/EEC, Directives 97/7/EC, 98/27/EC and 2002/65/EC of the European Parliament and of the Council and Regulation (EC) No 2006/2004 of the European Parliament and of the Council ('Unfair Commercial Practices Directive').

<sup>144</sup> <https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52013DC0196>

The minimum requirements and score for the verifier(s) as specified below shall be fulfilled. If the verification/validation is performed by a single verifier, he or she shall satisfy all the minimum requirements and the minimum score (see Section 8.3.1); if the verification/validation is performed by a team, the team as a whole shall satisfy all the minimum requirements and the minimum score. The documents proving the qualifications of the verifier(s) shall be provided as annex to the verification report or they shall be made available electronically.

In case a verification team is established, one of the members of the verification team shall be appointed as lead verifier.

### **8.3.1 Minimum requirements for verifier(s)**

The assessment of the competences of verifier or verification team is based on a scoring system that takes into account (i) verification and validation experience, (ii) LCA/EF methodology and practice, and (iii) knowledge of relevant technologies, processes or other activities included in the product(s)/organisation(s) in scope of the study. Table 29 presents the scoring system for each relevant competence and experience topic.

Unless otherwise specified in the context of the intended application, the verifier's self-declaration on the scoring system constitutes the minimum requirement. Verifier(s) shall provide a self-declaration of their qualifications (e.g. university diploma, working experience, certifications, etc.), stating how many points they achieved for each criterion and the total points achieved. This self-declaration shall form part of the LCA verification report.

A verification of a LCA study shall be conducted as per the requirements of the intended application. Unless otherwise specified, the minimum necessary score to qualify as a verifier or a verification team is six points, including at least one point for each of the three mandatory criteria (i.e. verification and validation practice, LCA/EF methodology and practice, and knowledge of technologies or other activities relevant to the LCA study).

**Table 29.** Scoring system for each relevant competence and experience topic for the assessment of the competences of verifier(s).

	Topic	Criteria	Score (points)				
			0	1	2	3	4
<b>Mandatory criteria</b>	Verification and validation practice	Years of experience <sup>(1)</sup>	<2	$2 \leq x < 4$	$4 \leq x < 8$	$8 \leq x < 14$	$\geq 14$
		Number of verifications <sup>(2)</sup>	$\leq 5$	$5 < x \leq 10$	$11 \leq x \leq 20$	$21 \leq x \leq 30$	$> 30$
	LCA/EF methodology and practice	Years of experience <sup>(3)</sup>	<2	$2 \leq x < 4$	$4 \leq x < 8$	$8 \leq x < 14$	$\geq 14$
		Number of LCA/EF studies or reviews <sup>(4)</sup>	$\leq 5$	$5 < x \leq 10$	$11 \leq x \leq 20$	$21 \leq x \leq 30$	$> 30$
	Knowledge of the specific sector	Years of experience <sup>(5)</sup>	<1	$1 \leq x < 3$	$3 \leq x < 6$	$6 \leq x < 10$	$\geq 10$
<b>Additional criteria</b>	Review, verification/ validation practice	Optional scores relating to verification/ validation	– 2 points: Accreditation as third party verifier for EMAS – 1 point: Accreditation as third party reviewer for at least one EPD Scheme, ISO 14001, or other EMS				

<sup>(1)</sup> Years of experience in the field of environmental verifications and/or review of LCA/PEF/EPD studies.

<sup>(2)</sup> Number of verifications for EMAS, ISO 14001, International EPD scheme or other EMS.

<sup>(3)</sup> Years of experience in the field of LCA modelling. Work done during master and bachelor degrees shall be excluded. Work done during a relevant Ph.D./Doctorate course shall be accounted for. Experience in LCA modelling includes, among others:

- LCA modelling in commercial and non-commercial software,
- Datasets and database development.

<sup>(4)</sup> Studies compliant with one of the following standards/methods: PEF, OEF, ISO 14040-44, ISO 14067, ISO 14025

<sup>(5)</sup> Years of experience in a sector related to the studied product(s). The experience in the sector may be gained through LCA studies or through other types of activities. The LCA studies shall be done on behalf of and with access to primary data of the producing/operating industry. The qualification of knowledge about technologies or other activities is assigned according to the classification of NACE codes (*Regulation (EC) No 1893/2006 of the European Parliament and of the Council of 20 December 2006 establishing the statistical classification of economic activities - NACE Revision 2*). Equivalent classifications of other international organisations may also be used. Experience gained with technologies or processes in a whole sector are considered valid for any of its sub-sectors.

### 8.3.2 Role of the lead verifier in the verification team

The lead verifier is a team member with additional tasks. The lead verifier shall:

- Distribute the tasks to be fulfilled between the team members according to the specific competencies of the team members, to get the full coverage of the tasks to be done and to use in the best manner the specific competencies of the team members;
- Coordinate the whole verification/validation process and ensure that all team members have a common understanding of the tasks they need to fulfil;
- Assemble all comments and ensure they are communicated to the commissioner of the LCA study in a clear and comprehensible way;
- Resolve any conflicting statements between team members;

- Ensure that the verification report and validation statement are generated and are signed by each member of the verification team.

## **8.4 Verification / validation requirements**

The verifier(s) shall describe all the outcomes related to the verification of the LCA study, the LCA report and the LCA communication vehicles, and give the commissioner of the LCA study the opportunity to improve the work, if necessary. Depending on the nature of the outcomes, additional iterations of comments and responses may be necessary. Any changes made in response to the verification outcomes shall be documented in the verification report.

The verification/validation shall be done by combining documental review and model validation.

- The documental review includes the LCA report, the technical content of any communication vehicle, and the data used in the calculations through requested underlying documents. Verifier(s) may organise the documental review either as an “at desk” or “on site” exercise, or as a mix of the two. The verification of the company-specific data shall always be organised through a visit of the production site(s) the data refer to.
- The validation of the model may take place at the production site of the commissioner of the study or be organised remotely. The verifier(s) shall access the model to verify its structure, the data used, and its consistency with the LCA report. The details about how the verifier(s) accesses the model shall be agreed by the commissioner of the LCA study and the verifier(s).

The verification may take place at the end of the LCA study or in parallel (concurrent) to the study.

The verifier(s) shall ensure that data verification/validation includes:

1. Coverage, precision, completeness, representativeness, consistency, reproducibility, sources and uncertainty;
2. Plausibility, quality and accuracy of the LCA-based data;
3. Quality and accuracy of additional environmental and technical information;
4. Quality and accuracy of the supporting information.

The validation of the LCA report shall be carried out by checking enough information to provide reasonable assurance that the LCA report fulfils all the conditions listed in Section 8.4.1.

The verification and validation of the LCA study shall be carried out by following the minimum requirements listed below (Sections 8.4.1 and 8.4.2).

### **8.4.1 Minimum requirements for the verification and validation of the PEF study**

The verifier(s) shall validate the accuracy and reliability of the quantitative information used in the calculation of the study. As this may be highly resource intensive, the following requirements shall be followed:

- The verifier shall check if the correct version of all impact assessment methods was used. For each of the most relevant impact categories (ICs), at least 50% of the characterisation shall be verified, while all normalisation and weighting factors of all ICs shall be verified. In particular, the verifier shall check that the characterisation factors correspond to those included in the impact assessment method the study declares compliance with<sup>145</sup>;

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<sup>145</sup> Available at: <http://eplca.jrc.ec.europa.eu/LCDN/developer.xhtml>.

- Cut-off applied (if any) fulfils the requirements at Section 4.6.4;
- All the newly created datasets shall be checked on their EF compliance (for the meaning of EF compliant datasets refer to <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>). All their underlying data (elementary flows, activity data and sub-processes) shall be validated;
- The aggregated EF compliant dataset of the product in scope is made available to the European Commission;
- For at least 70% of the most relevant processes (by number) in situation 2 option 2 of the DNM, 70% of the underlying numbers shall be validated. The 70% data shall include all energy and transport sub-processes for processes in situation 2 option 2;
- For at least 60% of the most relevant processes (by number) in situation 3 of the DNM, 60% of the underlying data shall be validated;
- For at least 50% of the other processes (by number) in situation 1, 2 and 3 of the DNM, 50% of the underlying data shall be validated.

The verifier shall put together in a single list all the most relevant processes coming from all the most relevant impact categories, together with their situation in the DNM.

For all processes to be validated, it shall be checked if the DQR satisfies the minimum DQR as specified in the *Plastics LCA* method (see Section 4.7.5).

These data checks shall include, but should not be limited to, the activity data used, the selection of secondary sub-processes, the selection of the direct elementary flows and the CFF parameters. For example, if there are 5 processes and each one of them includes 5 activity data, 5 secondary datasets and 10 CFF parameters, then the verifier(s) has to check at least 4 out of 5 processes (70%) and, for each process, (s)he shall check at least 4 activity data (70% of the total amount of activity data), 4 secondary datasets (70% of the total amount of secondary datasets), and 7 CFF parameters (70% of the total amount of CFF parameters), i.e. the 70% of each of data that could be possible subject of check.

#### **8.4.2 Verification and validation techniques**

The verifier shall assess and confirm whether the calculation methodologies applied are of acceptable accuracy, reliable, are appropriate and performed in accordance to the *Plastics LCA* method. The verifier shall confirm the correct application of conversion of measurement units.

The verifier shall check if applied sampling procedures are in accordance with the sampling procedure defined in the *Plastics LCA* method. The data reported shall be checked against the source documentation in order to check their consistency.

The verifiers shall evaluate whether the methods for making estimates are appropriate and have been applied consistently.

The verifier may assess alternatives to estimations or choices made, in the assertion to determine whether a conservative choice has been selected.

The verifier may identify uncertainties that are greater than expected and assess the effect of the identified uncertainty on the final LCA results.

#### **8.4.3 Data confidentiality**

Data for validation shall be presented in a systematic and comprehensive way, all the project documentation supporting the validation of a LCA study shall be provided to the verifier(s), including the LCA model, the confidential information and data. This data and information shall be treated as confidential and shall be used only during the verification process.

Confidential information may be excluded from the report, provided that:

- The request for non-disclosure only cover input information, not any output information;
- The commissioner of the LCA study provides the verifier with sufficient information on the nature of the data and information, and the reason for the request of excluding the data or information from the study report;
- The verifier accept the non-disclosure and include in the verification report the reasons for doing so;
- The commissioner of the LCA study keep a file of the non-disclosed information for possible future re-evaluation of the decision of non-disclosure.

Business data could be of confidential nature because of competition aspects, intellectual property rights or similar legal restrictions. Therefore, business data identified as confidential and provided during validation process shall be kept confidential. Hence, verifiers shall not disseminate or otherwise retain for use, without the permission of the organisation, any information disclosed to them during the course of the review work. The commissioner of the LCA study may ask to the verifier(s) to sign a non-disclosure agreement (NDA).

## **8.5 Outputs of the verification/ validation process**

### **8.5.1 Content of the verification and validation report**

The verification and validation report<sup>146</sup> shall include all findings of the verification/validation process, the actions taken by the commissioner to answer the comments of the verifier(s), and the final conclusion. The report is mandatory, but it may be confidential.

The final conclusion may be of different nature:

- “Compliant” if the documental or on-site information proves that the requirements of this section are fulfilled.
- “Not compliant” if the documental or on-site information proves that the requirements of this section are not fulfilled.
- “Complementary information needed” if the documental or on-site information cannot allow the verifier to conclude on compliance. This may happen if the information is not transparently or sufficiently documented or registered.

### **8.5.2 Content of the validation statement**

The validation statement is mandatory and shall always be provided as an annex to the LCA report. Therefore, from each communication vehicle it shall be possible to have access to the complete LCA report (except any confidential annexes), including the validation statement.

The following elements and aspects shall be included in the validation statement, as a minimum:

- Title of the LCA study under verification/validation, together with the exact version of the report to which the validation statement belongs;
- The commissioner of the LCA study;
- The user of the *Plastics LCA* method;
- The verifier(s) or, in the case of a verification team, the team members with the identification of the lead verifier;

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<sup>146</sup> The two aspects, validation and verification, are included in one report.

- Absence of conflicts of interest of the verifier(s) with respect to concerned products and any involvement in previous work (where relevant, PEFCR development, Technical Secretariat membership, consultancy work carried out for the user of the *Plastics LCA* method or of any applied PEFCR during the last three years);
- A description of the objective of the verification/validation;
- A statement of the result of the verification/validation;
- Any limitations of the verification/validation outcomes;
- Date in which the validation statement has been issued;
- Signature by the verifier(s).

### **8.5.3 Validity of the verification and validation report and the validation statement**

A verification/validation report and a validation statement shall refer only to one specific LCA report. The verification and validation report and a validation statement shall unambiguously identify the specific LCA study under verification (e.g. by including the title, the commissioner of the LCA study, the user of the *Plastics LCA* method, etc.), together with the explicit version of the final LCA report to which the verification and validation report and a validation statement apply (e.g. by including the report date, the version number, etc.).

Both the verification and validation report and the validation statement shall be completed on the basis of the final LCA report, after the implementation of all the corrective actions requested by the verifier(s). They shall carry the handwritten or electronic signature of the verifier(s).

The maximum validity of the verification and validation report and of the validation statement should not exceed three years starting from their first issue date.

During the validity period of the verification, surveillance (follow-up) shall be agreed between the commissioner of the LCA study and the verifier(s) to evaluate if the content is still consistent with the current situation (the suggested periodicity for this follow up is once per year).

The periodic checks shall focus on the parameters that according to the verifiers might lead to relevant changes in the results of the LCA study. A non-exhaustive list of such parameters is:

- Bill of Material / bill of components;
- Energy mix used for processes in situation 1 of the Data Needs Matrix;
- Change of packaging;
- Changes in the suppliers (materials/geography);
- Changes in the logistics;
- Relevant technological changes in the processes in situation 1 of the Data Needs Matrix.

At the time of the periodic check, the reasons for non-disclosure of information should also be reconsidered. The surveillance verification may be organised as a documental check and/or through on-site inspections.

Regardless of the validity, the LCA study (and consequently the LCA report) shall be updated during the surveillance period if the results of one of the impact categories communicated has worsened by more than 10.0% compared to the verified data, or if the total aggregated score has worsened by more than 5.0% compared to the verified data.

If these changes affect also the communication content, it shall be updated accordingly.

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## List of acronyms and abbreviations (used in the main text)<sup>147</sup>

<b>ABS</b>	Acrylonitrile Butadiene Styrene
<b>AD</b>	Activity Data
<b>ADEME</b>	Agence de l'Environnement et de la Maîtrise de l'Énergie
<b>ADP</b>	Abiotic Resource Depletion
<b>AE</b>	Accumulated Exceedance
<b>B2B</b>	Business to Business
<b>B2C</b>	Business to Consumer
<b>Bio-PBS</b>	Bio-based Polybutylene Succinate
<b>Bio-PE</b>	Bio-based Polyethylene
<b>Bio-PET</b>	Bio-based Polyethylene Terephthalate
<b>Bio-PP</b>	Bio-based Polypropylene
<b>BOD</b>	Biochemical Oxygen Demand
<b>BoM</b>	Bill of Materials
<b>BSI</b>	British Standards Institution
<b>CCU</b>	Carbon Capture and Utilisation
<b>CEN</b>	European Committee for Standardization
<b>CF</b>	Characterisation Factor
<b>CFCs</b>	Chlorofluorocarbons
<b>CFF</b>	Circular Footprint Formula
<b>COD</b>	Chemical Oxygen Demand
<b>CPA</b>	Classification of Products by Activity
<b>CTU</b>	Comparative Toxic Unit
<b>DAC</b>	Direct Air Capture
<b>DC</b>	Distribution centre
<b>dLUC</b>	Direct Land Use Change
<b>DM</b>	Dry Matter
<b>DNM</b>	Data Needs Matrix
<b>DQR</b>	Data Quality Rating
<b>DU</b>	Declared Unit
<b>EC</b>	European Commission
<b>EF</b>	Environmental Footprint
<b>EF</b>	Elementary Flow
<b>EIA</b>	Environmental Impact Assessments
<b>EMAS</b>	Eco-Management and Audit Scheme
<b>EMS</b>	Environmental Management System

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<sup>147</sup> Acronyms and abbreviations used in Annexes are defined throughout the respective text at their first occurrence. However, some of the acronyms and abbreviations used in such sections may be also reported in this list.

<b>EN</b>	European Norm
<b>EOl</b>	End of Life
<b>EPBP</b>	European PET Bottle Platform
<b>EPD</b>	Environmental Product Declaration
<b>EPS</b>	Expanded Polystyrene
<b>EU</b>	European Union
<b>FAO</b>	Food and Agriculture Organization of the United Nations
<b>FU</b>	Functional Unit
<b>GeR</b>	Geographical representativeness
<b>GHG</b>	Greenhouse Gas
<b>GWP</b>	Global warming potential
<b>HDPE</b>	High Density Polyethylene
<b>IC</b>	Impact Category
<b>IEC</b>	International Electrotechnical Commission
<b>ILCD</b>	International Reference Life Cycle Data System
<b>ILCD-EL</b>	International Reference Life Cycle Data System – Entry Level
<b>iLUC</b>	Indirect Land Use Change
<b>IPCC</b>	Intergovernmental Panel on Climate Change
<b>ISO</b>	International Organization for Standardization
<b>JRC</b>	Joint Research Centre
<b>LCA</b>	Life Cycle Assessment
<b>LCDN</b>	Life Cycle Data Network
<b>LCI</b>	Life Cycle Inventory
<b>LCIA</b>	Life Cycle Impact Assessment
<b>LCS</b>	Life Cycle Stage
<b>LDPE</b>	Low Density Polyethylene
<b>LEAP</b>	Livestock Environmental Assessment and Performance
<b>LHV</b>	Lower Heating Value
<b>LT</b>	Lifetime
<b>LULUC</b>	Land Use and Land Use Change
<b>NACE</b>	Nomenclature Générale des Activités Economiques dans les Communautés Européennes
<b>NDA</b>	Non-disclosure agreement
<b>NF</b>	Normalisation factor
<b>NMVOC</b>	Non-methane volatile compounds
<b>NPK</b>	Nitrogen-Phosphorus-Potassium
<b>ODP</b>	Ozone Depletion Potential
<b>OEF</b>	Organisation Environmental Footprint
<b>OEFSR</b>	Organisation Environmental Footprint Sector Rules

<b>P</b>	Precision
<b>PAS</b>	Publicly Available Specification
<b>PBS</b>	Polybutylene Succinate
<b>PC</b>	Polycarbonate
<b>PCR</b>	Product Category Rule
<b>PE</b>	Polyethylene
<b>PEF</b>	Polyethylene Furanoate
<b>PEF</b>	Product Environmental Footprint
<b>PEFCR</b>	Product Environmental Footprint Category Rule
<b>PET</b>	Polyethylene Terephthalate
<b>PLA</b>	Polylactic Acid
<b>PLP</b>	Plastic Leak Project
<b>PM</b>	Particulate Matter
<b>PP</b>	Polypropylene
<b>PS</b>	Polystyrene
<b>PUR</b>	Polyurethane
<b>PVC</b>	Polyvinylchloride
<b>TeR</b>	Technological representativeness
<b>TiR</b>	Time representativeness
<b>TKN</b>	Total Kjeldahl Nitrogen
<b>TOC</b>	Total Oxygen Demand
<b>TPS</b>	Thermoplastic Starch
<b>TR</b>	Technical Report
<b>TRL</b>	Technology Readiness Level
<b>TS</b>	Technical Secretariat
<b>UN</b>	United Nations
<b>UNEP</b>	United Nations Environment Programme
<b>UUID</b>	Universally Unique Identifier
<b>VS</b>	Volatile Solids
<b>WF</b>	Weighting Factor
<b>WMO</b>	World Meteorological Organisation
<b>WW</b>	Wet Weight

## List of definitions

**Acidification** – impact category that addresses impacts due to acidifying substances in the environment. Emissions of NO<sub>x</sub>, NH<sub>3</sub> and SO<sub>x</sub> lead to releases of hydrogen ions (H<sup>+</sup>) when the gases are mineralised. The protons contribute to the acidification of soils and water when they are released in areas where the buffering capacity is low, resulting in forest decline and lake acidification.

**Activity data** – This term refers to information which is associated with processes while modelling Life Cycle Inventories (LCI). The aggregated LCI results of the process chains that represent the activities of a process are each multiplied by the corresponding activity data<sup>148</sup> and then combined to derive the potential environmental impact associated with that process. Examples of activity data include quantity of kilowatt-hours of electricity used, quantity of fuel used, output of a process (e.g. waste), number of hours equipment is operated, distance travelled, floor area of a building, etc. Synonym of “non-elementary flow”.

**Additional Environmental Information** – Environmental information outside the default impact categories that is calculated and communicated alongside the LCA results.

**Additional Technical Information** – Non-environmental information that is calculated and communicated alongside the LCA results.

**Additive** – Substance added to a plastic polymer or product in order to modify its properties and to improve its performance (e.g. rigidity, flexibility, colour, durability etc.). Examples of additives include stabilisers, colorants, fillers, and plasticisers.

**Aggregated dataset** - Complete or partial life cycle of a product system that next to the elementary flows (and possibly not relevant amounts of waste flows and radioactive wastes) lists in the input/output list exclusively the product(s) of the process as reference flow(s), but no other goods or services. Aggregated datasets are also called “LCI results” datasets. The aggregated dataset may have been aggregated horizontally and/or vertically.

**Allocation** – An approach to solving multi-functionality problems. It refers to “partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems” (ISO 14040:2006).

**Application-specific** – It refers to the generic aspect of the specific application in which a material is used. For example, the average recycling rate of PET in bottles.

**Attributional** – Refers to process-based modelling intended to provide a static representation of average conditions, excluding market-mediated effects.

**Average data** – Refers to a production-weighted average of specific data.

**Background processes** – Refers to those processes in the product life cycle for which no direct access to information is possible. For example, most of the upstream life-cycle processes and generally all processes further downstream will be considered part of the background processes.

**Benchmark** – A standard or point of reference against which any comparison may be made. In the context of PEF and of the *Plastics LCA* method, the term ‘benchmark’ refers to the average environmental performance of the representative product sold in the EU market.

**Bill of Materials** – A bill of materials or product structure (sometimes bill of material, BOM or associated list) is a list of the raw materials, sub-assemblies, intermediate assemblies, sub-components, parts and the quantities of each needed to manufacture the product in scope of the LCA study. In some sectors it is equivalent to the bill of components.

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<sup>148</sup> Based on GHG protocol scope 3 definition from the [Corporate Accounting and Reporting Standard](#) (World resources institute, 2011).

**Bio-based plastic / polymer or plastic product** – A plastic, polymer or plastics product that is totally or partially derived from biomass.

**Biodegradable plastic / polymer or plastic product** – A plastic, polymer or plastic product capable of undergoing physical and biological decomposition by microorganisms, such that it ultimately decomposes, in the presence of oxygen, to carbon dioxide (CO<sub>2</sub>), water, mineral salts of any other elements present (mineralisation) and new biomass, or, in the absence of oxygen, to carbon dioxide, methane, mineral salts and new biomass. Biodegradable polymers and plastic products may be recoverable through composting and/or anaerobic digestion in accordance with European standards for packaging (EN 13432) or plastic materials (EN 14995).

**Biodegradation** – Intended as *Ultimate Biodegradation*, i.e. the complete breakdown of an organic chemical compound or organic material by microorganisms, in the presence of oxygen to CO<sub>2</sub>, water, mineral salts of any other element present (mineralisation) and new biomass, or, in the absence of oxygen, to CO<sub>2</sub>, methane, mineral salts and new biomass.

**Biodegradation rate (biodegradability, or “biodegradation percentage”)** – it has to be intended as the *carbon mineralisation rate*, unless otherwise specified<sup>149</sup>. It represents the share of total carbon in waste biodegradable polymers or plastic products which is converted (mineralised) to carbon dioxide (CO<sub>2</sub>) and possibly methane (CH<sub>4</sub>) in the specific conditions where biodegradability is claimed (e.g. controlled composting or anaerobic digestion conditions or in soil). The biodegradation rate is usually determined according to testing procedures recommended in different biodegradability standards, depending on the conditions where it needs to be proven, e.g. ISO 14855-1 (controlled composting conditions); ISO 15985 (high-solids anaerobic digestion conditions); ISO 17556 (in soil).

**Biomass** – Material produced by the growth of microorganisms, plants or animals.

**Bioplastic / Biopolymer or bioplastic product** – A plastic, polymer or plastic product that is totally or partially bio-based, biodegradable, or both.

**Business to Business (B2B)** – Describes transactions between businesses, such as between a manufacturer and a wholesaler, or between a wholesaler and a retailer.

**Business to Consumers (B2C)** – Describes transactions between business and consumers, such as between retailers and consumers. According to ISO 14025:2006, a consumer is defined as “an individual member of the general public purchasing or using goods, property or services for private purposes”.

**CO<sub>2</sub>-based plastic / polymer or plastic product** – A plastic, polymer or plastic product that is totally or partially derived from CO<sub>2</sub> captured from gaseous effluents or Direct Air Capture.

**Characterisation** – Calculation of the magnitude of the contribution of each classified input/output to their respective impact categories, and aggregation of contributions within each category. This requires a linear multiplication of the inventory data with characterisation factors for each substance and impact category of concern. For example, with respect to the impact category “Climate Change”, CO<sub>2</sub> is chosen as the reference substance and kg CO<sub>2</sub>-equivalents as the reference unit.

**Characterisation factor** – Factor derived from a characterisation model which is applied to convert an assigned Life Cycle Inventory result to the common unit of the impact category indicator (based on ISO 14040:2006).

**Classification** – Assigning the material/energy inputs and outputs tabulated in the Life Cycle Inventory to impact categories according to each substance’s potential to contribute to each of the impact categories considered.

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<sup>149</sup> Such as the biodegradation rate of Volatile Solids (VS) in the product/material or of the product/material as a whole.

**Climate Change** – All inputs or outputs that result in greenhouse gas emissions. The consequences include increased average global temperatures and sudden regional climatic changes. Climate change is an impact affecting the environment on a global scale.

**Co-function** – Any of two or more functions resulting from the same unit process or product system.

**Commissioner of the LCA study** – Organisation (or group of organisations) that finances the LCA study in accordance with the *Plastics LCA* method and a relevant PEFCR, if available (definition adapted from ISO 14071/2014, point 3.4).

**Company-specific data** – It refers to directly measured or collected data from one or multiple facilities (site-specific data) that are representative for the activities of the company. It is synonymous to “primary data”. To determine the level of representativeness a sampling procedure may be applied.

**Company-specific dataset** – It refers to a dataset (disaggregated or aggregated) compiled with company-specific data. In most cases the activity data is company-specific while the underlying sub-processes are datasets derived from background databases.

**Comparative assertion** – An environmental claim regarding the superiority or equivalence of one product versus a competing product that performs the same function (including the benchmark of the product category) (adapted from ISO 14044:2006).

**Comparison** – A comparison, not including a comparative assertion, (graphic or otherwise) of two or more products based on the results of a LCA study and supporting PEFCRs (if available).

**Co-product** – Any of two or more products resulting from the same unit process or product system (ISO 14040:2006).

**Cradle to Gate** – A partial product supply chain, from the extraction of raw materials (cradle) up to the manufacturer’s “gate”. The distribution, storage, use stage and end of life stage of the supply chain are omitted.

**Cradle to Grave** – A product’s life cycle that includes raw material extraction, processing, distribution, storage, use, and disposal or recycling stages. All relevant inputs and outputs are considered for all of the stages of the life cycle.

**Data quality** – Characteristics of data that relate to their ability to satisfy stated requirements (ISO 14040:2006). Data quality covers various aspects, such as technological, geographical and time-related representativeness, as well as completeness and precision of the inventory data.

**Data Quality Rating (DQR)** – Semi-quantitative assessment of the quality criteria of a dataset based on Technological representativeness, Geographical representativeness, Time-related representativeness, and Precision. The data quality shall be considered as the quality of the dataset as documented.

**Delayed emissions** – Emissions that are released over time, e.g. through long-use or final disposal stages, compared to a single emission at time t closer to uptake from biomass or resource extraction and product manufacturing.

**Direct elementary flows** (also named elementary flows) – All output emissions and input resource use that arise directly in the context of a process. Examples are emissions from a chemical process, or fugitive emissions from a boiler directly onsite.

**Direct land use change (dLUC)** – The transformation from one land use type into another, which takes place in a unique land area and does not lead to, or consider, any potential land use changes in other systems.

**Directly attributable** – Refers to a process, activity or impact occurring within the defined system boundary.

**Disaggregation** – The process that breaks down an aggregated dataset into smaller unit process datasets (horizontal or vertical). The disaggregation may help making data more specific. The process of disaggregation should never compromise or threaten to compromise the quality and consistency of the original aggregated dataset

**Downstream** – Occurring along a product supply chain after the point of referral.

**Ecotoxicity - freshwater** – impact category that addresses the toxic impacts on an ecosystem, which damage individual species and change the structure and function of the ecosystem. Ecotoxicity is a result of a variety of different toxicological mechanisms caused by the release of substances with a direct effect on the health of the ecosystem.

**EF compliant dataset** – Dataset developed in compliance with the EF requirements provided at <http://eplca.jrc.ec.europa.eu/LCDN/developer.xhtml>.

**Elementary flows** – In the Life Cycle Inventory, elementary flows include “material or energy entering the system being studied that has been drawn from the environment without previous human transformation, or material or energy leaving the system being studied that is released into the environment without subsequent human transformation” (ISO 14040, 3.12). Elementary flows include, for example, resources taken from nature or emissions into air, water, soil that are directly linked to the characterisation factors of the default impact categories.

**Environmental aspect** – Element of an organisation’s activities or products or services that interacts or can interact with the environment (ISO 14001:2015).

**Environmental impact** – Any change to the environment, whether adverse or beneficial, that wholly or partially results from an organisation’s activities, products or services (EMAS regulation).

**Environmental mechanism** – System of physical, chemical and biological processes for a given impact category linking the Life Cycle Inventory results to impact category indicators (based on ISO 14040:2006).

**Eutrophication** – Nutrients (mainly nitrogen and phosphorus) from sewage outfalls and fertilised farmland accelerate the growth of algae and other vegetation in water. The degradation of organic material consumes oxygen resulting in oxygen deficiency and, in some cases, fish death. Eutrophication translates the quantity of substances emitted into a common measure expressed as the oxygen required for the degradation of dead biomass. Three impact categories are used in the *Plastics LCA* method to assess the impacts due to eutrophication: Eutrophication - terrestrial; Eutrophication - freshwater; Eutrophication - marine.

**External communication** – Communication to any interested party other than the commissioner or the practitioner of the study.

**Extrapolated data** – Refers to data from a given process that is used to represent a similar process for which data is not available, on the assumption that it is reasonably representative.

**Flow diagram** – Schematic representation of the flows occurring during one or more process stages within the life cycle of the product being assessed.

**Foreground elementary flows** – Direct elementary flows (emissions and resources) for which access to primary data (or company-specific information) is available.

**Foreground processes** – Refer to those processes in the product life cycle for which direct access to information is available. For example, the producer’s site and other processes operated by the producer or its contractors (e.g. goods transport, head-office services, etc.) belong to the foreground processes.

**Fossil-based plastic / polymer or plastic product** – A plastic, polymer or plastic product that is totally derived from fossil resources such as oil and natural gas.

**Fragmentation** – The process by which plastic products break into smaller pieces over time. A plastic can fragment into microscopic pieces while not being biodegradable.

**Functional unit** – The functional unit defines the qualitative and quantitative aspects of the function(s) and/or service(s) provided by the product being evaluated. The functional unit definition answers the questions “what?”, “how much?”, “how well?”, “for how long?”, “where”, and “for whom”.

**Gate to Gate** – A partial product supply chain that includes only the processes carried out on a product within a specific organisation or site.

**Gate to Grave** – A partial product supply chain that includes only the distribution, storage, use, and disposal or recycling stages.

**Global Warming Potential (GWP)** – Capacity of a greenhouse gas to influence radiative forcing, expressed in terms of a reference substance (for example, CO<sub>2</sub>-equivalent units) and specified time horizon (e.g. GWP 20, GWP 100, GWP 500, for 20, 100, and 500 years respectively). It relates to the capacity to influence changes in the global average surface-air temperature and subsequent change in various climate parameters and their effects, such as storm frequency and intensity, rainfall intensity and frequency of flooding, etc.

**Horizontal averaging** – it is the action of aggregating multiple unit process datasets or aggregated process datasets in which each provides the same reference flow in order to create a new process dataset (UN Environment, 2011).

**Human Toxicity - cancer** – impact category that accounts for adverse health effects on human beings caused by the intake of toxic substances through inhalation of air, food/water ingestion, penetration through the skin insofar as they are related to cancer.

**Human Toxicity – non-cancer** – impact category that accounts for the adverse health effects on human beings caused by the intake of toxic substances through inhalation of air, food/water ingestion, penetration through the skin insofar as they are related to non-cancer effects that are not caused by particulate matter/respiratory inorganics or ionising radiation.

**Impact category** – Class of resource use or environmental impact to which the Life Cycle Inventory data are related.

**Impact category indicator** – Quantifiable representation of an impact category (based on ISO 14000:2006).

**Independent external expert** – Competent person, not employed in a full-time or part-time role by the commissioner of the LCA study or the user of the *Plastics LCA* method, and not involved in defining the scope or conducting the LCA study (adapted from ISO 14071/2014, point 3.2).

**Indirect land use change (iLUC)** – It occurs when a demand for a certain land use or land-based commodity leads to changes outside the system boundary, i.e. in other land use types. These indirect effects may be mainly assessed by means of economic modelling of the demand for land or by modelling the relocation of activities on a global scale.

**Input flows** – Product, material or energy flow that enters a unit process. Products and materials include raw materials, intermediate products and co-products (ISO 14040:2006).

**Intermediate product** – Output from a unit process that is input to other unit processes that require further transformation within the system (ISO 14040:2006). An intermediate product is a product that requires further processing before it is saleable to the final consumer.

**Ionising Radiation - human health** – impact category that accounts for the adverse health effects on human health caused by radioactive releases.

**Land Use** – impact category related to use (occupation) and conversion (transformation) of land area by activities such as agriculture, forestry, roads, housing, mining, etc. Land occupation considers the effects of the land use, the amount of area involved and the duration of its occupation (changes in quality multiplied by area and duration). Land transformation considers the extent of changes in land properties and the area affected (changes in quality multiplied by the area).

**LCA communication vehicles** – It includes all the possible ways that may be used to communicate the results of the LCA study to the stakeholders (e.g. labels, environmental product declarations, green claims, websites, infographics, etc.).

**LCA profile** – The quantified results of a LCA study. It includes the quantification of the impacts for the various impact categories and the additional environmental information considered necessary to report. Synonym of Environmental Profile.

**LCA report** – Document that summarises the results of the LCA study.

**LCA study** – Term used to identify the totality of actions needed to calculate the LCA results. It includes the modelling, the data collection, and the analysis of the results. It excludes the LCA report and the verification of the LCA study and report.

**Lead verifier** – Verifier taking part in a verification team with additional responsibilities compared to the other verifiers in the team.

**Life cycle** – Consecutive and interlinked stages of a product system, from raw material acquisition or generation from natural resources to final disposal (ISO 14040:2006).

**Life cycle approach** – Takes into consideration the spectrum of resource flows and environmental interventions associated with a product from a supply-chain perspective, including all stages from raw material acquisition through processing, distribution, use, and end of life processes, and all relevant related environmental impacts (instead of focusing on a single issue).

**Life Cycle Assessment (LCA)** – Compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle (ISO 14040:2006).

**Life Cycle Impact Assessment (LCIA)** – Phase of the LCA study aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts for a product system throughout the life cycle of the product (based on ISO 14044:2006). The default Life Cycle Impact Assessment methods adopted in the *Plastics LCA* provide impact characterisation factors for elementary flows in order to aggregate the impact to obtain a limited number of midpoint indicators.

**Life Cycle Impact Assessment (LCIA) method** – Protocol for quantitative translation of Life Cycle Inventory data into contributions to an environmental impact of concern.

**Life Cycle Inventory (LCI)** – The combined set of exchanges of elementary, waste and product flows in a LCI dataset.

**Life Cycle Inventory (LCI) dataset** – A document or file with life cycle information of a specified product or other reference (e.g., site, process), covering descriptive metadata and quantitative Life Cycle Inventory. A LCI dataset could be a unit process dataset, partially aggregated or an aggregated dataset.

**Litter** – Regardless of the size, any persistent, manufactured or processed solid material discarded, disposed of or abandoned improperly, without consent, at an inappropriate location.

**Littering** – Discarding, disposing of or abandoning improperly any persistent, manufactured or processed solid material, without consent, at an inappropriate location.

**Loading rate** – Ratio of actual load to the full load or capacity (e.g. mass or volume) that a vehicle carries per trip.

**Macro-plastics** – solid, synthetic, polymer-containing particles greater than 5 mm in their longest dimension, and which may contain additives or other substances.

**Material-specific** – It refers to a generic aspect of a material. For example, the recycling rate of PET.

**Micro-plastics** – solid, synthetic, polymer-containing particles of no more than five millimetres in their longest dimension (i.e. with a diameter  $\Phi \leq 5$  mm).

**Mineralisation rate** – share of total carbon in waste biodegradable polymers or plastic products which is converted (gasified) to carbon dioxide (CO<sub>2</sub>) and possibly methane (CH<sub>4</sub>) in the specific conditions where biodegradability is claimed (e.g. controlled composting or anaerobic digestion conditions or in soil). The mineralisation rate is usually determined according to testing procedures recommended in different biodegradability standards, depending on the conditions where it needs to be proven, e.g. ISO 14855-1 (controlled composting conditions); ISO 15985 (high-solids anaerobic digestion conditions); ISO 17556 (in soil). Synonym of “biodegradability” and “biodegradation percentage”.

**Monomer** – A substance which is capable of forming covalent bonds with a sequence of additional like or unlike molecules under the conditions of the relevant polymer-forming reaction used for the particular process (Regulation (EC) No 1907/2006)<sup>150</sup>. In other words, a monomer is a molecule making up the smallest repeating unit in a polymer. Monomers undergo chemical conversion to form the bonds holding them together in a polymer.

**Multi-functionality** – If a process or facility provides more than one function, i.e. it delivers several goods and/or services (“co-products”), then it is “multifunctional”. In these situations, all inputs and emissions linked to the process will be partitioned between the product of interest and the other co-products according to clearly stated procedures.

**Nano-plastics** – solid, synthetic, polymer-containing particles of no more than 1 micrometre in their longest dimension (i.e. with a diameter  $\Phi \leq 1$   $\mu$ m).

**Non-elementary (or complex) flows** – In the Life Cycle Inventory, non-elementary flows include all the inputs (e.g. electricity, materials, transport processes) and outputs (e.g. waste, by-products) in a system that need further modelling efforts to be transformed into elementary flows. Synonym of “activity data”.

**Normalisation** – After the characterisation step, normalisation is the step in which the Life Cycle Impact Assessment results are multiplied by normalisation factors that represent the overall inventory of a reference unit (e.g. a whole country or an average citizen). Normalised Life Cycle Impact Assessment results express the relative shares of the impacts of the analysed system in terms of the total contributions to each impact category per reference unit. When displaying the normalised Life Cycle Impact Assessment results of the different impact topics next to each other, it becomes evident which impact categories are affected most and least by the analysed system. Normalised Life Cycle Impact Assessment results reflect only the contribution of the analysed system to the total impact potential, not the severity/relevance of the respective total impact. Normalised results are dimensionless, but not additive.

**Output flows** – Product, material or energy flow that leaves a unit process. Products and materials include raw materials, intermediate products, co-products and releases (ISO 14040:2006).

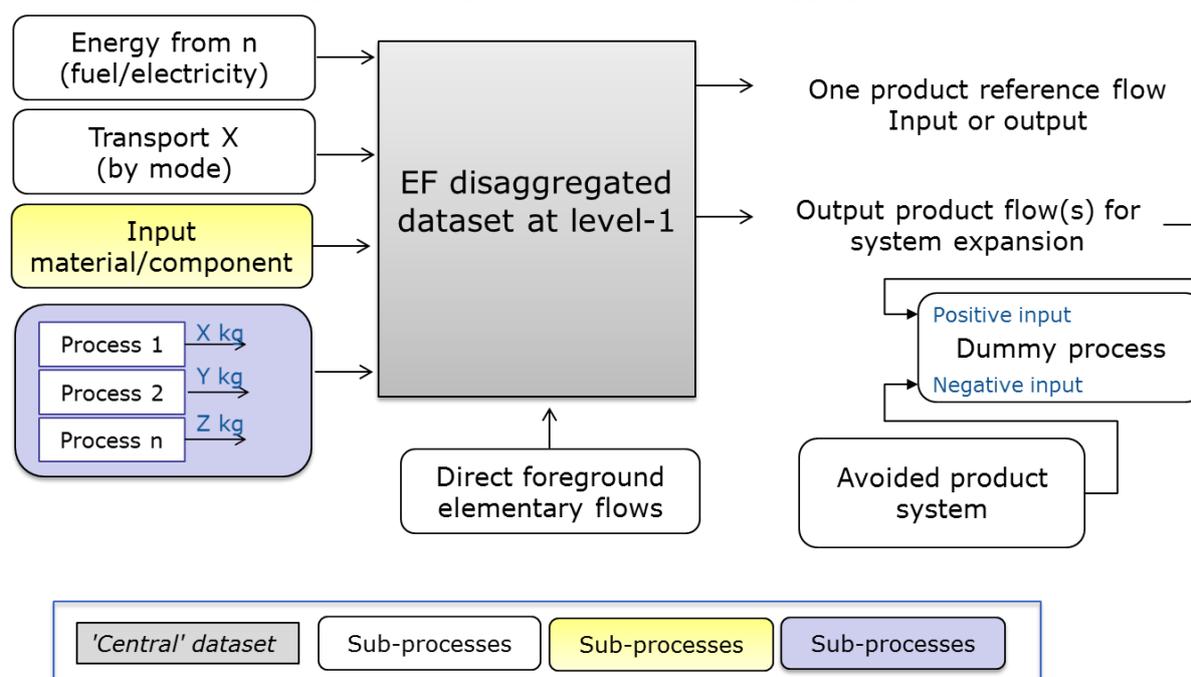
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<sup>150</sup> Regulation (EC) No 1907/2006 of the European Parliament and of the Council of 18 December 2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH), establishing a European Chemicals Agency, amending Directive 1999/45/EC and repealing Council Regulation (EEC) No 793/93 and Commission Regulation (EC) No 1488/94 as well as Council Directive 76/769/EEC and Commission Directives 91/155/EEC, 93/67/EEC, 93/105/EC and 2000/21/EC (OJ L 396, 30.12.2006, p. 1).

**Ozone Depletion** – impact category that accounts for the degradation of stratospheric ozone due to emissions of ozone-depleting substances, for example long-lived chlorine and bromine containing gases (e.g. CFCs, HCFCs, Halons).

**Partially disaggregated dataset** – A dataset with a LCI that contains elementary flows and activity data, and that only in combination with its complementing underlying datasets yield a complete aggregated LCI data set.

**Partially disaggregated dataset at level-1** – A partially disaggregated dataset at level-1 contains elementary flows and activity data of one level down in the supply chain, while all complementing underlying datasets are in their aggregated form.



**Figure 15.** Example of dataset partially disaggregated at Level-1.

**Particulate Matter** – impact category that accounts for the adverse health effects on human health caused by emissions of Particulate Matter (PM) and its precursors (NO<sub>x</sub>, SO<sub>x</sub>, NH<sub>3</sub>).

**Photochemical Ozone Formation** – impact category that accounts for the formation of ozone at the ground level of the troposphere caused by photochemical oxidation of volatile organic compounds (VOCs) and carbon monoxide (CO) in the presence of nitrogen oxides (NO<sub>x</sub>) and sunlight. High concentrations of ground-level tropospheric ozone damage vegetation, human respiratory tracts and manmade materials through reaction with organic materials.

**Plastic** – A material consisting of a polymer as defined in point 5 of Article 3 of Regulation (EC) No 1907/2006, to which additives or other substances may have been added, and which can function as a main structural component of final products, with the exception of natural polymers that have not been chemically modified (conforming to Directive 2019/904 EU)<sup>151</sup>.

**Polymer** – A substance consisting of molecules characterised by the sequence of one or more types of monomer units. Such molecules must be distributed over a range of molecular weights wherein differences in the molecular weight are primarily attributable to differences in the number of monomer units. A polymer comprises the following: (a) a simple weight majority of molecules containing at least three monomer units which are

<sup>151</sup> Directive (EU) 2019/904 of the European Parliament and of the Council of 5 June 2019 on the reduction of the impact of certain plastic products on the environment.

covalently bound to at least one other monomer unit or other reactant; (b) less than a simple weight majority of molecules of the same molecular weight (Definition according to Regulation (EC) No 1907/2006)<sup>152</sup>. In other words, a polymer is a substance consisting of molecules characterised by the sequence of one or more types of monomers.

**Population** – Any finite or infinite aggregation of individuals, not necessarily animate, subject to a statistical study.

**Primary data**<sup>153</sup> – This term refers to data from specific processes within the supply chain of the user of the *Plastics LCA* method or user of a specific PEFCR. Such data may take the form of activity data, or foreground elementary flows (Life Cycle Inventory). Primary data are site-specific, company-specific (if multiple sites for the same product) or supply chain specific. Primary data may be obtained through meter readings, purchase records, utility bills, engineering models, direct monitoring, material/product balances, stoichiometry, or other methods for obtaining data from specific processes in the value chain of the user of the *Plastics LCA* method or user of a specific PEFCR. In this method, primary data is synonym of "company-specific data" or "supply-chain specific data".

**Product** – Any goods or services (ISO 14040:2006).

**Product category** – Group of products (or services) that can fulfil equivalent functions (ISO 14025:2006).

**Product Category Rules (PCRs)** – Set of specific rules, requirements and guidelines for developing Type III environmental declarations for one or more product categories (ISO 14025:2006).

**Product Environmental Footprint Category Rules (PEFCRs)** – Product category specific, life cycle based rules that complement general methodological guidance for PEF/*Plastics LCA* studies by providing further specification at the level of a specific product category. PEFCRs help to shift the focus of the PEF/*Plastics LCA* study towards those aspects and parameters that matter the most, and hence contribute to increased relevance, reproducibility and consistency of the results by reducing costs versus a study based on the comprehensive requirements of the PEF/*Plastics LCA* method. Only the PEFCRs listed on the European Commission website ([http://ec.europa.eu/environment/eussd/smcp/PEFCR\\_OEFSR\\_en.htm](http://ec.europa.eu/environment/eussd/smcp/PEFCR_OEFSR_en.htm)) are recognised as in line with these methods.

**Product flow** – Products entering from or leaving to another product system (ISO 14040:2006).

**Product system** – Collection of unit processes with elementary and product flows, performing one or more defined functions, and which models the life cycle of a product (ISO 14040:2006).

**Raw material** – Primary or secondary material that is used to produce a product (ISO 14040:2006).

**Reference flow** – Measure of the outputs from processes in a given product system required to fulfil the function expressed by the functional unit (based on ISO 14040:2006).

**Refurbishment** – It is the process of restoring components to a functional and/ or satisfactory state to the original specification (providing the same function), using methods such as resurfacing, repainting, etc. Refurbished products may have been tested and verified to function properly.

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<sup>152</sup> Regulation (EC) No 1907/2006 of the European Parliament and of the Council of 18 December 2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH), establishing a European Chemicals Agency, amending Directive 1999/45/EC and repealing Council Regulation (EEC) No 793/93 and Commission Regulation (EC) No 1488/94 as well as Council Directive 76/769/EEC and Commission Directives 91/155/EEC, 93/67/EEC, 93/105/EC and 2000/21/EC (OJ L 396, 30.12.2006, p. 1).

<sup>153</sup> Based on GHG protocol scope 3 definition from the [Corporate Accounting and Reporting Standard](#) (World resources institute, 20011).

**Releases** – Emissions to air and discharges to water and soil (ISO 14040:2006).

**Representative product (model)** – The RP may be a real or a virtual (non-existing) product. The virtual product should be calculated based on average European market sales-weighted characteristics of all existing technologies/materials covered by the product category or sub-category. Other weighting sets may be used, if justified, for example weighted average based on mass (ton of material) or weighted average based on product units (pieces).

**Representative sample** – A representative sample with respect to one or more variables is a sample in which the distribution of these variables is exactly the same (or similar) as in the population from which the sample is a subset.

**Resource Use - fossils** – impact category that addresses the use of non-renewable fossil natural resources (e.g. natural gas, coal, oil).

**Resource Use - minerals and metals** – impact category that addresses the use of non-renewable abiotic natural resources (minerals and metals).

**Sample** – A sample is a subset containing the characteristics of a larger population. Samples are used in statistical testing when population sizes are too large for the test to include all possible members or observations. A sample should represent the whole population and not reflect bias toward a specific attribute.

**Secondary data**<sup>154</sup> – It refers to data not from a specific process within the supply-chain of the company performing a LCA study. This refers to data that is not directly collected, measured, or estimated by the company, but sourced from a third party LCI database or other sources. Secondary data includes industry average data (e.g., from published production data, government statistics, and industry associations), literature studies, engineering studies and patents, and may also be based on financial data, and contain proxy data, and other generic data. Primary data that go through a horizontal aggregation step are considered as secondary data.

**Secondary micro-plastics** – Micro-plastics originating from the fragmentation and/or degradation of larger plastic items or products (macro-plastics) into smaller plastic particles once released into the environment (terrestrial, riverine or marine). Micro-plastics generation takes place through photodegradation, oxidation, and other weathering processes.

**Sensitivity analysis** – Systematic procedures for estimating the effects of the choices made regarding methods and data on the results of a LCA study (based on ISO 14040: 2006).

**Site-specific data** – It refers to directly measured or collected data from one facility (production site). It is synonymous to “primary data”.

**Specific data** – Refers to directly measured or collected data representative of activities at a specific facility or set of facilities. Synonymous with “primary data.”

**Subdivision** – Subdivision refers to disaggregating multifunctional processes or facilities to isolate the input flows directly associated with each process or facility output. The process is investigated to see whether it may be subdivided. Where subdivision is possible, inventory data should be collected only for those unit processes directly attributable to the products/services of concern.

**Sub-population** – Any finite or infinite aggregation of individuals, not necessarily animate, subject to a statistical study that constitutes a homogenous sub-set of the whole population. Synonymous with “stratum”.

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<sup>154</sup> Based on GHG protocol scope 3 definition from the [Corporate Accounting and Reporting Standard](#) (World resources institute, 2011)

**Sub-processes** – Those processes used to represent the activities of the level 1 processes (=building blocks). Sub-processes may be presented in their (partially) aggregated form (see Figure 15).

**Sub-sample** – A sample of a sub-population.

**Supply chain** – It refers to all of the upstream and downstream activities associated with the operations of the user of the *Plastics LCA* method, including the use of sold products by consumers and the end of life treatment of sold products after consumer use.

**Supply chain specific** – It refers to a specific aspect of the specific supply chain of a company. For example the recycled content value of an aluminium may be produced by a specific company.

**System boundary** – Definition of aspects included or excluded from the study. For example, for a “cradle-to-grave” LCA study, the system boundary includes all activities from the extraction of raw materials through the processing, distribution, storage, use, and disposal or recycling stages.

**System boundary diagram** – Graphic representation of the system boundary defined for the LCA study.

**Temporary carbon storage** – happens when a product “reduces the GHGs in the atmosphere” or “creates negative emissions”, by removing carbon from the atmosphere and storing it for a limited amount of time.

**Type III environmental declaration** – An environmental declaration providing quantified environmental data using predetermined parameters and, where relevant, additional environmental information (ISO 14025:2006). The predetermined parameters are based on the ISO 14040 series of standards, which is made up of ISO 14040 and ISO 14044.

**Uncertainty analysis** – Procedure to assess the uncertainty in the results of a LCA study due to data variability and choice-related uncertainty.

**Unit process** – Smallest element considered in the LCI for which input and output data are quantified (based on ISO 14040:2006).

**Unit process, black box** – Process chain or plant level unit process. This covers horizontally averaged unit processes across different sites. Covers also those multi-functional unit processes, where the different co-products undergo different processing steps within the black box, hence causing allocation problems for this dataset.

**Unit process, single operation** – Unit operation type unit process that cannot be further subdivided. Covers multi-functional processes of unit operation type.

**Upstream** – Occurring along the supply chain of purchased goods/ services prior to entering the system boundary.

**User of the *Plastics LCA* method** – a stakeholder producing a LCA study based on the *Plastics LCA* method.

**User of the *Plastics LCA* results** – a stakeholder using the results of *Plastics LCA* study for any internal or external purpose.

**Validation** – Confirmation by the verifier, that the information and data included in the LCA study, the LCA report and the communication vehicles are reliable, credible and correct.

**Validation statement** – Conclusive document aggregating the conclusions from the verifiers or the verification team regarding the LCA study. This document is mandatory and shall carry the electronic or handwritten signature of the verifier or, in case of a verification panel, of the lead verifier.

**Verification** – Conformity assessment process carried out by a verifier to demonstrate whether the LCA study has been carried out in compliance with the most updated version of the LCA method adopted by the Commission.

**Verification report** – Documentation of the verification process and findings, including detailed comments from the verifier(s), as well as the corresponding responses. This document is mandatory, but it may be confidential. The document shall carry the electronic or handwritten signature of the verifier, or in case of a verification panel, of the lead verifier.

**Verification team** – Team of verifiers that will perform the verification of the LCA study, of the LCA report, and of the LCA communication vehicles.

**Verifier** – Independent external expert performing a verification of the LCA study and eventually taking part in a verification team.

**Vertical aggregation** – Technical- or engineering-based aggregation refers to vertical aggregation of unit processes that are directly linked within a single facility or process train. Vertical aggregation involves combining unit process datasets (or aggregated process datasets) together linked by a flow (UN Environment, 2011).

**Waste** – Substances or objects which the holder intends or is required to dispose of (ISO 14040:2006).

**Water Use** – It represents the relative available water remaining per area in a watershed, after the demand of humans and aquatic ecosystems has been met. It assesses the potential of water deprivation, to either humans or ecosystems, building on the assumption that the less water remaining available per area, the more likely another user will be deprived (see also <http://www.wulca-waterlca.org/aware.html>).

**Weighting** – Weighting is a step that supports the interpretation and communication of the results of the analysis. LCA results are multiplied by a set of weighting factors, which reflect the perceived relative importance of the impact categories considered. Weighted LCA results may be directly compared across impact categories, and also summed across impact categories to obtain a single overall score.

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## **Annexes**

## Annex A – Differences compared to the PEF method

Table A.1 provides a detailed overview of the main specific differences between the *Plastics LCA* method and the PEF method, distinguished into the three main following categories:

1. Additional provisions or information: including additional requirements or recommendations on methodological, modelling or reporting aspects that are not directly addressed in the PEF method (but defined in total agreement with its overarching rules and principles). This category also includes additional descriptions or discussion of a specific methodological or modelling issue (or even other general issues) to facilitate understanding by users of this method.
2. More specific provisions or information: including specific requirements, recommendations and/or information that further specify, with no contradiction, those already provided in the PEF method for a given methodological/modelling or reporting aspect. Additional or more specific/relevant examples provided in the *Plastics LCA* method for such aspects are also included in this category.
3. Structure differences: including identical provisions and/or information as those provided in the PEF method, while reported in different positions within the document (e.g. in different sections or in a different order within a same section).

Note that, while being as comprehensive as possible, this overview may not be totally exhaustive.

**Table A.1.** Summary of the main specific differences between the *Plastics LCA* method and the PEF method, according to the three main categories described in the text above (excluding spelling issues). Notes: (1) page numbers refer to the beginning of the relevant section where the difference occurs; (2) Differences in document structure (situation 3) are separately and progressively reported at the bottom of the table (while differences referring to situations 1 and 2 are reported together and still progressively).

Specific issue / difference	Page	Category		
		1	2	3
Provision of general introductory information on the <i>Plastics LCA</i> method	12	x		
Description of the context where the <i>Plastics LCA</i> method has been developed and its overall aim	13	x		
Description of the objective (and scope) of the <i>Plastics LCA</i> method	13		x	
Description of the development process of the <i>Plastics LCA</i> method	14	x		
Description of the target audience of the <i>Plastics LCA</i> method (reported independently)	15		x	
Description of the relationship of the <i>Plastics LCA</i> method with other methods and standards	15		x	
Description of the main general additions and changes included in the <i>Plastics LCA</i> method compared to the PEF method	16		x	

Description of available documents	18	x		
Adapted description of the phases of a <i>Plastics LCA</i> study	21		x	
Further specification of the role of PEFCRs and of their relation with the <i>Plastics LCA</i> method	23		x	
Description of anticipated perspectives for the <i>Plastics LCA</i> method	24	x		
Provision of a plastic-specific example of goal definition	25		x	
Provisions for a proper description and reporting of the product(s) in scope, of its technical characteristics and of its relevant function(s)	26	x		
Provisions for the selection of relevant product function(s) and on additional/secondary functions	28	x		
Further specification and exemplification of rules related to the definition of the functional unit (particularly also including the additional "where" and "for whom" aspects, and an expanded description of the "how well" and "how long" aspects)	30		x	
Further specification and exemplification of rules related to the calculation of the reference flow (including a specific focus on food packaging)	32		x	
Provision of plastic-specific examples on the definition of the functional unit and the associated reference flow	33		x	
System boundary diagram: inclusion of further specified provisions and of a plastic-specific example	35		x	
Description of the main potential "indirect effects" associated to plastics value chains, and clarification on their exclusion from the system boundary of <i>Plastics LCA</i> studies	38	x		
Inclusion of a broadened and more plastic-specific description of "additional information" relevant to <i>Plastics LCA</i> studies	42		X	
Definition of the specific type of "additional environmental information" that shall/should/may be included in a <i>Plastics LCA</i> study (and of the related calculation method, where relevant)	44		X	
Definition of the specific type of "additional technical information" that shall/may be included in a <i>Plastics LCA</i> study	46		x	
Limitations and assumptions: inclusion of further specified provisions and of (plastic-relevant) examples	47		x	
Inclusion of a figure clarifying the concept of two-step procedure that should be applied to compile the Life Cycle Inventory	49		x	

Provision of additional information on how to use the results from the screening step	49		x	
Further specification/exemplification of processes and activities that shall be included in the <i>Raw Material Acquisition and Pre-processing</i> stage (as far as relevant) depending on the type of feedstock used for plastic production (fossil-based resources, plastic waste, primary biomass, bio-based waste/by-products, and CO <sub>2</sub> captured from gaseous effluents)	50-52		x	
Further specification/exemplification of processes and activities that shall be included in the <i>Manufacturing</i> stage, as far as relevant	53		x	
Further specification/exemplification of processes and activities that shall be included in the <i>Distribution</i> stage (as far as relevant), and of rules on the handling and modelling of losses/waste of product, packaging and packaged products (e.g. food) occurring during this stage	53		x	
Further specification/exemplification of use-stage activities and input or output flows that shall be included, as far as relevant, in the <i>Use</i> stage (including a dedicated overview figure), and of rules on the handling and modelling of losses/waste of product, packaging and packaged products (e.g. food) occurring during this stage	55		x	
Further specification/exemplification of processes and activities to be included in the <i>End of Life</i> stage, and of related general modelling provisions (further developed in other subsequent sections)	58		x	
Provision of general modelling requirements on fossil-based feedstock supply	60	x		
Provision of general modelling requirements on the use of plastic waste as a feedstock	62	x		
Inclusion of a detailed list of activities, inputs and emissions that shall be considered, where relevant, in the modelling of agricultural production (and description of relevant parameters typically required for this purpose)	63	x		
Inclusion (under Section 4.4.3.4) of a specific requirement on how to determine the total land use duration for cultivated crops (periods between different crop-cycles are also to be accounted)	64		x	
Refined description of modelling requirements for fertiliser emissions and restriction of models applicable to calculate NO <sub>3</sub> -N emissions from leaching (only the alternative approach specified in the PEF method is allowed, with some adaptations)	65		x	
Refined description of modelling requirements for pesticides and related emissions	68		x	

Refined description of modelling requirements for heavy metal emissions and specification of a preferential modelling approach applicable in case of crops acting as a heavy metal sink (second option listed in the PEF method, corresponding to option #1 in the <i>Plastics LCA</i> method)	69		x	
Provision of default values to calculate CO <sub>2</sub> emissions from drained peat soils	69		x	
Refined description of modelling requirements for agricultural operations	70		x	
Provision of general modelling requirements on the use of bio-based waste and by-products as a feedstock	70	x		
Provision of general and more specific modelling requirements on the use of CO <sub>2</sub> captured from gaseous effluents as a feedstock	71	x		
Provision of general recommendations on the handling of plastic products relying on new, emerging, or maturing technologies	74	x		
Definition of general provisions on the modelling of the Use stage for plastic products and related examples	79		x	
Further specification of general requirements on End of Life modelling	82		x	
Inclusion of general provisions to define appropriate average End of Life scenarios for the product in scope (including non-commercially available products)	82	x		
Refined description and restriction of rules on the modelling of pre-consumer waste/scrap under the CFF (one of the two options allowed in the PEF method is specifically recommended, with scrap not being considered as recycled content)	89		x	
Further specification of rules on how to apply the CFF to compost and anaerobic digestion/sewage treatment (particularly on the inclusion of potential benefits from fertiliser substitution)	93		x	
Inclusion of examples on how to apply the CFF to plastic products (considering both a partially and a completely open-loop recycling situation)	94		x	
Provision of general and more specific rules to model relevant End of Life options for plastic products, including mechanical recycling, biological treatments (industrial composting and anaerobic digestion), on-land application of organic material from biological treatments, in-situ biodegradation (on/into the soil), incineration, landfilling, and release of plastic products in the environment	98-122	x		
Provision of specific rules on how to model the use of secondary energy and fuels under the CFF	126		x	

Inclusion of a general provision on the modelling of capital goods (if included in the LCA model)	126		x	
Refined description and restriction of rules on the modelling of biogenic GHG emissions and removals (all emissions and removals shall be modelled in the LCI; the simplified modelling approach allowed in the PEF method is excluded)	128		x	
Refined description and further specification of rules to quantify GHG emissions from direct Land Use Change and changes in soil carbon stocks	129		x	
Specification of rules to quantify GHG emissions from indirect Land Use Change (iLUC) for the purpose of providing "additional environmental information"	129	x		
Description of and brief discussion on temporary carbon storage and delayed carbon emissions, including related modelling provisions	132		x	
Refined description of rules to handle multi-functional processes	134		x	
Refined description of data collection requirements	135-137		x	
Refined description of data quality assessment and quality requirements	138-147		x	
Further specification of modelling rules on End of Life options that may be assumed for product losses/waste at distribution centres/retailers, during transport and at consumer	203		x	
Discussion on the relevance of potential indirect effects from fossil-based feedstock supply or displacement with alternative feedstocks	217	x		
Provision of background considerations and additional recommendations on the modelling of the use of captured CO <sub>2</sub> as a feedstock for plastic products	221	x		
Provision of general considerations on the assessment of products based on maturing or emerging technologies	227	x		
Presentation of an overview of available methods and approaches to quantify macro- and micro-plastics generation and release throughout the product life cycle (including product litter), and operational description of the <i>PLP method</i> that may be applied to provide "additional environmental information"	231	x		
Provision of background information on land use changes and on models available to quantify GHG emissions from iLUC in LCA	249	x		
Description of a systematic review of selected LCA studies on plastic products and related precursors	259	x		
<b>Main differences in document structure (1,2)</b>				

"Target audience" is reported as an independent section rather than as a combined "Objectives and target audience" section	15			x
"General requirements for Plastics LCA studies" is presented as an independent section rather than as part of the "Objectives and target audience" section	20			x
"Terminology used: shall, should, may" is reported in Section 2 (General considerations and requirements for Plastics LCA studies), rather than as independent initial section following the abstract	20			x
"Product Environmental Footprint Category Rules (PEFCRs) and relation with the Plastics LCA method" is reported in Section 2 (General considerations and requirements for Plastics LCA studies) rather than in Section 1 (Introduction)	23			x
Modelling requirements for agricultural production: the section on modelling of fertiliser emissions ("Fertilisers") is reported before that on the modelling of pesticide emissions ("Pesticides") (rather than the opposite)	65-68			x
The section "Agricultural operations" replaces the section "Other activities" reported in the PEF method	70			x
Modelling requirements on "Electricity use" are reported after those related to "Extended product lifetime", rather than after modelling requirements on "Agricultural production"	126			x
Modelling requirements on "Capital goods - infrastructures and equipment" are reported after those related to "Electricity use" (in the new position; see above), rather than after modelling requirements on "Transport and logistics"	126			x
Provisions on the "Sampling procedure" are reported after those related to "Capital goods" (in the new position; see above), rather than after modelling requirement on "Storage at distribution centre or retail"	126			x

(<sup>1</sup>) Changes in the sequence of sub-sections within a same overall main section of the compared documents are not taken into account (e.g. any changes of second-level sections within Sections 1 and 2).

(<sup>2</sup>) Differences are described considering the situation in the *Plastics LCA* method and the PEF method as a reference for comparison.

## Annex B - Default loss rates per type of product during the Distribution stage and at consumer (Use stage)

**Table B.1.** Default loss rates per type of product during the Distribution stage and at consumer (Use stage). Values are based on assumptions, if not specified otherwise. Out of simplification, the values for restaurant are considered the same as for consumer at home.

<b>Retail trade sector</b>	<b>Category</b>	<b>Loss rate (incl. broken products but not products returned to manufacturer) during Distribution (overall consolidated value for transportation, storage and retail place)</b>	<b>Loss rate at consumer – Use stage (including restaurant, etc.)</b>
<i>Food</i>	<i>Fruits and vegetables</i>	<i>10% (FAO, 2011)</i>	<i>19% (FAO, 2011)</i>
	<i>Meat and meat alternatives</i>	<i>4% (FAO, 2011)</i>	<i>11% (FAO, 2011)</i>
	<i>Dairy products</i>	<i>0.5% (FAO, 2011)</i>	<i>7% (FAO, 2011)</i>
	<i>Grain products</i>	<i>2% (FAO, 2011)</i>	<i>25% (FAO, 2011)</i>
	<i>Oils and fats</i>	<i>1% (FAO, 2011)</i>	<i>4% (FAO, 2011)</i>
	<i>Prepared/processed meals (ambient)</i>	<i>10%</i>	<i>10%</i>
	<i>Prepared/processed meals (chilled)</i>	<i>5%</i>	<i>5%</i>
	<i>Prepared/processed meals (frozen)</i>	<i>0.6% (primary data based on Picard – oral communication from Arnaud Brulaire)</i>	<i>0.5% (primary data based on Picard – oral communication from Arnaud Brulaire)</i>
	<i>Confectionery</i>	<i>5%</i>	<i>2%</i>
	<i>Other foods</i>	<i>1%</i>	<i>2%</i>
<i>Beverages</i>	<i>Coffee and tea</i>	<i>1%</i>	<i>5%</i>
	<i>Alcoholic beverages</i>	<i>1%</i>	<i>5%</i>
	<i>Other beverages</i>	<i>1%</i>	<i>5%</i>

<b>Retail trade sector</b>	<b>Category</b>	<b>Loss rate (incl. broken products but not products returned to manufacturer) during Distribution (overall consolidated value for transportation, storage and retail place)</b>	<b>Loss rate at consumer – Use stage (including restaurant, etc.)</b>
	<i>Tobacco</i>	0%	0%
	<i>Pet food</i>	5%	5%
	<i>Live animals</i>	0%	0%
	<i>Clothing and textile</i>	10%	0%
	<i>Footwear and leather goods</i>	0%	0%
	<i>Personal accessories</i>	0%	0%
<i>Home and professional supplies</i>	<i>Home hardware supplies</i>	1%	0%
	<i>Furniture, furnishings and decor</i>	0%	0%
	<i>Electrical household appliances</i>	1%	0%
	<i>Kitchen merchandise</i>	0%	0%
	<i>Information and communication equipment</i>	1%	0%
	<i>Office machinery and supplies</i>	1%	0%
<i>Cultural and recreational goods</i>	<i>Books, newspapers and paper/paper supplies</i>	1%	0%
	<i>Music and videos</i>	1%	0%
	<i>Sporting equipment and gadgets</i>	0%	0%

<b>Retail trade sector</b>	<b>Category</b>	<b>Loss rate (incl. broken products but not products returned to manufacturer) during Distribution (overall consolidated value for transportation, storage and retail place)</b>	<b>Loss rate at consumer – Use stage (including restaurant, etc.)</b>
	<i>Other cultural and recreational goods</i>	1%	0%
<i>Healthcare</i>		5%	5%
<i>Cleaning/hygiene products, cosmetics and toiletries</i>		5%	5%
<i>Fuels, gases, lubricants and oils</i>		1%	0%
<i>Batteries and power</i>		0%	0%
<i>Plants and garden supplies</i>	<i>Flowers, plants and seeds</i>	10%	0%
	<i>Other garden supplies</i>	1%	0%
<i>Other goods</i>		0%	0%
<i>Gas station</i>	<i>Gas station products</i>	1%	0%

**Food losses at distribution centre, during transport and at retail place, and at home:** may be assumed to be 50% disposed (i.e., incinerated and landfilled), 25% sent to composting, and 25% to anaerobic digestion, although supply-chain and context-specific data are preferable.

**Product losses (excluding food losses) during packing/ repacking/ unpacking at distribution centre, during transport and at retail place:** may be assumed to be 100% recycled, although supply-chain and context-specific data are preferable.

**Other waste generated at distribution centre, during transport and at retailer (except food and product losses) such as other waste from repacking/unpacking:** may be assumed to follow the same End of Life fate as for household waste, although supply-chain and context-specific data are preferable.

**Liquid food wastes (as for instance milk) at consumer (including restaurant, etc.):** may be assumed to be poured in the sink and therefore treated in wastewater treatment plants.

## **B.1. References**

FAO (2011). Global food losses and food waste – Extent, causes and prevention. FAO, Rome, Italy.

## **Annex C - List of default values for CFF parameters (A, R<sub>1</sub>, R<sub>2</sub>, R<sub>3</sub> and Q<sub>s</sub>/Q<sub>p</sub>)**

Annex C is available at <https://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml>.

The list of default values in Annex C is periodically reviewed and updated by the European Commission. Users of this method are invited to check and use the most updated values available in the Annex.

## Annex D - Default data for modelling the Use stage

The data and assumptions reported in Tables D.1 and D.2 shall be used to model Use stage activities, unless better data are available (which shall be made transparent and justified in the LCA study report).

**Table D.1.** Default data and assumptions to model crosscutting Use stage activities for several product categories (data are based on assumptions, unless specified otherwise).

Product	Use stage assumptions per product category
Meat, fish, eggs	<i>Chilled storage. Cooking: 10 minutes in frying pan (75% on gas and 25% electricity), 5 gram sunflower oil (incl. its life cycle) per kg product. Dishwashing of frying pan</i>
Milk	<i>Chilled storage, drunk cold in 200 ml glass (i.e., 5 glasses per L milk), incl. glass life cycle and dishwashing</i>
Pasta	<i>Per kg pasta cooked in pot with 10 kg water, 10 min boiling (75% on gas and 25% electricity). Boiling phase: 0.18 kWh per kg of water, Cooking phase: 0.05 kWh per minute of cooking</i>
Frozen dishes	<i>Frozen storage. Cooked in oven 15 minutes at 200°C (incl. a fraction of a stove, a fraction of a baking sheet). Baking sheet rinsing: 5 L water</i>
Roast and ground coffee	<i>7 g roast and ground coffee per cup Filter coffee preparation in a filter coffee machine: machine production and end-of-life (1.2 kg, 4380 uses, with 2 cups/use), paper filter (2 g/use), electricity consumption (33 Wh/cup) and water consumption (120 ml/cup). Machine rinsing/washing: 1 L cold water per use, 2 L hot water per 7 uses, decanter dishwashing (every 7 uses) Cup (mug) production and end-of-life and dishwashing Source: based on PEFCR Coffee (draft as of Feb 1, 2015)<sup>155</sup></i>
Beer	<i>Cooling, drunk in 33 cl glass (i.e., 3 glasses per L beer), glass production, end-of-life and dishwashing. See also PEFCR of beer<sup>156</sup></i>
Bottled water	<i>Chilled storage. Storage duration: 1 day. 2.7 glasses per L water drunk, 260 gram glass production, end-of-life and dishwashing</i>
Pet food	<i>Pet food dish production, end-of-life and dishwashing</i>
Goldfish	<i>Electricity and water use and treatment for the aquarium (43 kWh and 468 L per year). Goldfish feed production (1 g/day, assumed 50% fish meal, 50% soybean meal). Lifetime of the goldfish assumed to be 7.5 years</i>
T-shirt	<i>Washing machine, tumble dryer use and ironing. 52 washing at 41 degree, 5.2 tumble drying (10%) and 30 times ironing per T-shirt.</i>

<sup>155</sup> <https://webgate.ec.europa.eu/fpfis/wikis/display/EUENVFP/PEFCR+Pilot%3A+Coffee>.

<sup>156</sup> <http://ec.europa.eu/environment/eussd/smgp/pdf/Beer%20PEFCR%20June%202018%20final.pdf>.

<b>Product</b>	<b>Use stage assumptions per product category</b>
	<p><i>Washing machine: 70 kg, 50% steel, 35% plastic, 5% glass, 5% aluminium, 4% copper, 1% electronics, 1560 cycles (=loads) within its lifetime. 179 kWh and 8700 L water for 220 cycles at 8 kg load (based on <a href="http://www.bosch-home.com/ch/fr/produits/laver-et-s%C3%A9cher/lave-linge/WAQ28320FF.html?source=browse">http://www.bosch-home.com/ch/fr/produits/laver-et-s%C3%A9cher/lave-linge/WAQ28320FF.html?source=browse</a>) being 0.81 kWh and 39.5 L/cycle, as well as 70 ml laundry detergent/cycle</i></p> <p><i>Tumble dryer: 56 kg, same composition share and lifetime as for washing machine assumed. 2.07 kWh/cycle for 8 kg clothes load</i></p>
<i>Paint</i>	<i>Paint brush production, sand paper, ... (see PEFCR of decorative paints)<sup>157</sup></i>
<i>Cell Phone</i>	<i>kWh/year for the charge, 2 years lifetime</i>
<i>Laundry detergent</i>	<i>Use of a washing machine (see T-shirt data for washing machine model). 70 ml laundry detergent assumed per cycle, i.e., 14 cycles per kg detergent</i>
<i>Automotive oil</i>	<i>10% losses during use assessed as hydrocarbons emissions to water</i>

**Table D.2.** Default data and assumptions to model storage during the use stage (data are based on assumptions, unless specified otherwise).

<b>Product</b>	<b>Assumptions common to several product categories</b>
<i>Ambient storage (at home)</i>	<i>Ambient storage at home is considered, for the sake of simplification, as having no impact</i>
<i>Chilled storage (in a fridge, at home)</i>	<p><i>Storage time: product dependent. As default 7 days storage in fridge (ANIA and ADEME, 2012)</i></p> <p><i>Storage volume: assumed to be 3x the actual product volume</i></p> <p><i>Energy consumption: 0.0037 kWh/L (i.e., "the storage volume") - day (ANIA and ADEME, 2012)</i></p> <p><i>Fridge production and end-of-life considered (assuming 15 years of lifetime)</i></p>
<i>Chilled storage (at the pub/restaurant)</i>	<p><i>The fridge at the pub is assumed to consume 1400 kWh/ yr (Communication with Heineken green cooling expert, 2015). 100% of this energy consumption is assumed to be for the cooling of beer. The throughput of the fridge is assumed to be 40hl/ yr. This means 0.035 kWh/ l for pub / supermarket cooling for the full storage time</i></p> <p><i>Fridge production and end-of-life considered (assuming 15 years of lifetime)</i></p>

<sup>157</sup> [http://ec.europa.eu/environment/eussd/smgp/documents/PEFCR\\_decorative\\_paints.pdf](http://ec.europa.eu/environment/eussd/smgp/documents/PEFCR_decorative_paints.pdf).

<b>Product</b>	<b>Assumptions common to several product categories</b>
<i>Frozen storage (in a freezer, at home)</i>	<p><i>Storage time: 30 days in freezer (based on ANIA and ADEME, 2012)</i></p> <p><i>Storage volume: assumed to be 2x the actual product volume</i></p> <p><i>Energy consumption: 0.0049 kWh/L (i.e., "the storage volume") - day (ANIA and ADEME, 2012)</i></p> <p><i>Freezer production and end-of-life considered (assuming 15 years of lifetime): assumed similar to fridge</i></p>
<i>Cooking (at home)</i>	<p><i>Cooking: 1 kWh/h use (derived from consumptions for induction stove (0.588 kWh/h), ceramic stove (0.999 kWh/h) and electric stove (1.161 kWh/h) all from (ANIA and ADEME, 2012)</i></p> <p><i>Baking in oven: electricity considered: 1.23 kWh/h (ANIA and ADEME, 2012)</i></p>
<i>Dishwashing (at home)</i>	<p><i>Dishwasher use: 15 L water, 10 g soap and 1.2 kWh per washing cycle (Kaenzig and Jolliet, 2006).</i></p> <p><i>Dishwasher production and end-of-life considered (assuming 1500 cycle per lifetime).</i></p> <p><i>When dishwashing is done by hand, one assumes an equivalent of 0.5 L of water and 1 g of soap for the value above of 2.5% (with a scaling in terms of water use and soap, using the % above). The water is assumed to be warmed by natural gas, considering a delta T of 40 °C and an efficiency of energy from natural gas heating to water heat of 1/1.25 (meaning that to heat the 0.5 L of water one needs to use <math>1.25 * 0.5 * 4186 * 40 = 0.1</math> MJ of "Heat, natural gas, at boiler").</i></p>

## **D.1. References**

ANIA and ADEME (2012). *Projet de référentiel transversal d'évaluation de l'impact environnemental des produits alimentaires (mainly annexe 4) (« GT1 »)*, 23/04/12.

Kaenzig, J., Jolliet, O. (2006). *Consommation respectueuse de l'environnement: décisions et acteurs clés, modèles de consommation. Connaissance de l'environnement no 0616*. Berne, Switzerland: Office fédéral de l'environnement; 2006.

## **Annex E - LCA report template**

This Annex presents the template that shall be applied for LCA studies developed according to the *Plastics LCA* method. The template specifies the mandatory report structure to be followed and the information to be reported as a non-exhaustive list. All items required to be reported by the *Plastics LCA* method shall be included, even if they are not explicitly mentioned in this template.

# **LCA Report**

[Insert product name here]

## **Table of contents**

### **Acronyms**

[List in this section all the acronyms used in the LCA study. Those already included in the latest version of the *Plastics LCA* method shall be copied in their original form. The acronyms shall be provided in alphabetical order.]

### **Definitions**

[List in this section all the definitions that are relevant for the LCA study. Those already included in the latest version of the *Plastics LCA* method shall be copied in their original form. The definitions shall be provided in alphabetical order.]

### **E.1. SUMMARY**

[The summary shall include, as a minimum, the following elements:

- The goal and scope of the study, including relevant limitations and assumptions;
- A short description of the system boundary;
- Relevant statements about data quality;
- The main results of the LCIA: these shall be presented showing the results of all impact categories (characterized, normalized, weighted);
- A description of what has been achieved by the study, any recommendation made and conclusions drawn.

To the extent possible, the summary should be written with a non-technical audience in mind and should not be longer than 3-4 pages.]

### **E.2. GENERAL**

[The information below should ideally be placed on the front-page of the study:

- Name of the product (including a photo);
- Product identification (e.g. model number);
- Product classification (CPA) based on the latest CPA list version available,
- Company presentation (name, geographic location);
- Date of publication of the LCA study (the date shall be written in extended format, e.g. 25 June 2015, to avoid confusion over the date format);
- Geographic validity of the LCA study (countries where the product is consumed/sold);
- Compliance with the present method;
- Conformance to other documents, additional to the *present* method;
- Name and affiliation of the verifier(s).]

### **E.3. GOAL OF THE STUDY**

[Mandatory reporting elements include, as a minimum:

- Intended application(s);
- Methodological limitations;
- Reasons for carrying out the study;
- Target audience;
- Commissioner of the study;
- Identification of the verifier.]

### **E.4. SCOPE OF THE STUDY**

[The scope of the study shall identify the analysed system in detail and address the overall approach used to establish: i) functional unit and reference flow, ii) system boundary, iii) list of impact categories, iv) additional information (environmental and technical), v) assumptions and limitations.]

#### **E.4.1. Functional/declared unit and reference flow**

[Provide the functional unit, defining the four aspects:

- The function(s)/service(s) provided: **“what”**;
- The extent of the function or service: **“how much”**;
- The expected level of quality: **“how well”**;
- The duration/life time of the product: **“how long”**;
- The location/geography where the function or service is provided: **“where”**;
- The beneficiary of the function or service, e.g. a consumer/citizen, a professional, a pet animal, etc.: **“for whom”**.

Provide the declared unit, in case the functional unit cannot be defined (e.g. if the product in scope is an intermediate product).

Provide the reference flow.]

#### **E.4.2. System boundary**

[This section shall include as a minimum:

- All life-cycle stages that are part of the product system. In case the naming of the default life cycle stages has been changed, the user shall specify to which default life cycle stage it corresponds. Document and justify if life cycle stages were split and/or new ones were added.
- The main processes covered in each life cycle stage (details are in the LCI section E.5). The co-products, by-products and waste streams of at least the foreground system shall be clearly identified.
- The reason for and potential significance of any exclusion.
- A system boundary diagram with the processes that are included and those excluded, highlighting those activities which falls respectively under situation 1, 2, and 3 of the Data Needs Matrix, and highlighting where company-specific data are used.]

### **E.4.3. Impact categories**

[Provide a table with the list of default impact categories, units, and EF reference package used (see <http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml> for further details).

For Climate Change, specify if the results of the three sub-indicators are reported separately in the results section.]

### **E.4.4. Additional information**

[Describe any additional environmental information and additional technical information included in the LCA study. Provide references and exact calculations rules adopted.

Explain if biodiversity is relevant/not relevant for the product in scope.

When the product in scope is an intermediate product, additional technical information shall include:

- The biogenic and separately the non-biogenic (e.g. fossil) carbon content at factory gate (physical content and allocated content).
- Recycled content ( $R_1$ ).
- Results with application-specific A-values of the Circular Footprint Formula, if relevant.]

### **E.4.5. Assumptions and limitations**

[Describe all limitations and assumptions. Provide list of data gaps, if any, and the way in which these gaps were possibly filled (through EF compliant proxy datasets or ILCD-EL compliant datasets). Provide list of proxy datasets used.]

## **E.5. LIFE CYCLE INVENTORY ANALYSIS**

[This section shall describe the compilation of the Life Cycle Inventory (LCI) and include:

- Screening step, if performed;
- List and description of life cycle stages;
- Description of modelling choices;
- Description of allocation approaches applied;
- Description and documentation of data used and sources;
- Data quality requirements and rating.]

### **E.5.1. Screening step [if applicable]**

[Provide a description of the screening step, including relevant information regarding data collection, data used (e.g. list of secondary data sets, activity data, direct elementary flows), cut-off, and results of the Life Cycle Impact Assessment phase.

Document the main findings and any refinement of the initial scope settings (if any).]

### **E.5.2. Modelling choices**

[Describe all modelling choices for the applicable aspects listed below (more can be added, when relevant):

- Agricultural production;

- Transport and logistics: all data used shall be provided in the LCA report (e.g. transport distances, payloads, re-use rates for packaging, etc.). If default scenarios were not used in the modelling, provide documentation of all specific data used;
- Capital goods: if capital goods are included, the LCA report shall include a clear and extensive explanation, reporting all assumptions made;
- Storage and retail;
- Use stage: Product dependent processes shall be included in the system boundary of the LCA study. Product independent processes shall be excluded from the system boundary and qualitative information may be provided, see Section 4.4.9 of the present method. Describe the approach taken to model the Use stage (main function approach or delta approach);
- End of life modelling, including values of parameters of the Circular Footprint Formula ( $A$ ,  $B$ ,  $R_1$ ,  $R_2$ ,  $Q_s/Q_p$ ,  $R_3$ ,  $LHV$ ,  $X_{ER,heat}$ ,  $X_{ER,elec}$ ), list of processes and datasets used ( $E_v$ ,  $E_{rec}$ ,  $E_{recEoL}$ ,  $E^*_v$ ,  $E_d$ ,  $E_{ER}$ ,  $E_{SE,heat}$ ,  $E_{SE,elec}$ ) with reference to Annex C of the present method;
- Extended product lifetime;
- Electricity use;
- Sampling procedure (report if a sampling procedure was applied and indicate the approach taken);
- Greenhouse gas emissions and removals;
- Offsets (if reported as additional environmental information).]

### **E.5.3. Handling multi-functional processes**

[Describe the allocation rules used in the LCA study and how the modelling/calculations were made. Provide the list of all allocation factors used for each process and the detailed list of processes and datasets used, in case substitution is applied.]

### **E.5.4. Data collection**

[This section shall include as a minimum:

- Description and documentation of all company-specific data collected:
  - List of processes covered by company-specific data indicating to which life cycle stage they belong;
  - List of resource use and emissions (i.e. direct elementary flows);
  - List of activity data used;
  - Link to detailed bill of materials and/or ingredients, including substance names, units and quantities, including information on grades/purities and other technically and/or environmentally relevant characterisation of these;
  - Company-specific data collection/estimation/calculation procedures;
- List of all secondary datasets used (process name, UUID, dataset source (node on Life Cycle Data Network, data stock) and compliance with the latest EF compliant data sets requirements and reference package);
- Modelling parameters;
- Cut-off applied, if any;
- Sources of published literature;

- Validation of data, including documentation;
- If a sensitivity analysis has been conducted, this shall be reported.]

#### **E.5.5. Data quality requirements and rating**

[Provide a table listing all processes and their situation according to the Data Needs Matrix (DNM).

Provide the DQR of the LCA study.]

### **E.6. IMPACT ASSESSMENT RESULTS [CONFIDENTIAL, IF RELEVANT]**

#### **E.6.1. LCA results**

[This section shall include as a minimum:

- Characterised results of all impact categories shall be calculated and reported as absolute values in the LCA report. The sub-categories 'Climate Change – fossil', 'Climate Change – biogenic' and 'Climate Change - land use and land use change', shall be reported separately if they show a contribution of more than 5% each to the total score of Climate Change);
- Normalised and weighted results as absolute values (for each impact category);
- Weighted results as single score;
- Results of the Use stage for final products shall be reported separately.]

#### **E.6.2. Additional information**

[This section shall include:

- Results of the additional environmental information;
- Results of the additional technical information.]

### **E.7. INTERPRETATION OF LCA RESULTS**

[This section shall include as a minimum:

- Assessment of the robustness of the LCA study;
- List of most relevant impact categories, life cycle stages, processes and elementary flows (see the example table below);
- Limitations and relationship of the LCA results relative to the defined goal and scope of the LCA study;
- Conclusions, recommendations, limitations and improvement potentials).]

**Example:** exemplary table for reporting identified most relevant impact categories, life cycle stages, processes and elementary flows.

Most relevant impact category	[%]	Most relevant life cycle stage	[%]	Most relevant processes	[%]	Most relevant elementary flows	[%]
IC 1		End of life		Process 1		el. flow 1	
						el. flow 2	
				Process 2		el. flow 2	
		Raw material acquisition and pre-processing		Process 4		el. flow 1	
IC 2		Manufacturing		Process 1		el. flow 2	
						el. flow 3	
IC 3		Manufacturing		Process 1		el. flow 2	
						el. flow 3	

## E.8. VALIDATION STATEMENT

[The validation statement is mandatory and shall always be provided as public annex of the public LCA report.

The following elements and aspects shall be included in the validation statement, as a minimum:

- Title of the LCA study under verification/validation, together with the exact version of the report to which the validation statement belongs;
- The commissioner of the LCA study;
- The user(s) of the present method / practitioner(s) who has/have implemented the LCA study;
- The verifier(s) or, in the case of a verification team, the team members with the identification of the lead verifier;
- Absence of conflicts of interest of the verifier(s) with respect to concerned products and any involvement in previous work (where relevant, PEFCR development, Technical Secretariat membership, consultancy work carried out for the user of the present method or of any applied PEFCR during the last three years);
- A description of the objective of the verification/validation;

- A statement of the result of the verification/validation;
- Any limitations of the verification/validation outcomes;
- Date in which the validation statement has been issued;
- Signature by the verifier(s).]

### **ANNEX I**

[The Annex serves to document supporting elements to the main report which are of a more technical nature. It could include:

- Bibliographic references;
- Detailed Life Cycle Inventory analysis (optional if considered sensitive and communicated separately in the confidential annex, see below);
- Detailed assessment of data quality: Provide i) Data Quality Rating per process in accordance with the present method and ii) Data Quality Rating for the newly created EF compliant datasets. In case information is confidential, it shall be included in Annex II.]

### **ANNEX II – CONFIDENTIAL REPORT**

[The Confidential annex is an optional chapter that shall contain all those data (including raw data) and information that are confidential or proprietary and cannot be made externally available.]

### **ANNEX III – EF COMPLIANT DATASET**

[The aggregated EF-compliant dataset of the product in scope shall be made available to the European Commission.]

## **Annex F - Discussion on the relevance of potential indirect effects from fossil-based feedstock supply or displacement with alternative feedstocks**

A number of potential indirect effects are discussed in the literature in relation to fossil-based feedstock supply for transportation fuels (Unnasch et al., 2009) or their displacement with alternative feedstocks. As reported in Section 3.2.3.2, such effects may include:

- iLUC caused by agricultural expansion on afforested areas due to road construction on previously occupied agricultural land for accessing oil fields;
- Emissions and impacts associated with military operations required to protect petroleum supply, as well as impacts from military conflicts to secure access to oil resources (and possible need for reconstruction);
- Effects potentially induced by possible changes in production and subsequent market availability of refinery co-products, due to a reduced fossil fuel demand to refineries when alternative fuels are used. This would lead to decreased crude oil processing and subsequent reduced production and availability of refinery outputs<sup>158</sup>. In the case of replacing fossil-based feedstocks for polymers with alternative ones, such effects may be generated by a reduced demand for naphtha rather than fuels;
- Macro-economic effects due to changes in petroleum usage and price (also referred to as “rebound effect”).

In addition to these effects, Malins et al. (2015) consider accidents, including oil spills and oil fires, as potential indirect effects. However, these can be rather considered direct unintended effects related to fossil-based feedstock supply, as better discussed below.

Assessing such indirect effects is challenging and may involve many uncertainties (Malins et al. 2015). For example, it is not straightforward to determine which portion of deforestation caused by road expansion on occupied agricultural land for oil field accessing is additional to what would have occurred anyway. Unnasch et al. (2009) estimated this impact to be 0.6-1.0 g CO<sub>2</sub> eq. per MJ of fossil transport fuel produced in Ecuador. However, taking into account the global share of petroleum supply from Ecuador, the overall impact due to additional deforestation in such country would be less than 0.01 g CO<sub>2</sub> eq. per MJ of fuel supplied worldwide. According to Malins et al. (2015), when around 20% of global oil production comes from nations with large areas covered by tropical forests, the actual total carbon implication of road-induced deforestation is almost certainly no more than 0.1 g CO<sub>2</sub> eq./MJ. In addition, it has to be noted that also biofuel supply can involve similar effects, such as in Indonesia and Malaysia, where road infrastructure develops around the growing palm oil industry, potentially leading to further deforestation.

Estimating impacts from military protection of petroleum supply is also challenging, due to incomplete public data on military operations that are actually conducted, and to uncertainty on which ones are directly related to protecting petroleum supply. In addition, there is no consensus on the extent to which guaranteeing oil supply might have been a primary reason for conflicts (e.g. in Iraq). An attempt to quantify these emissions has been conducted, for instance, by Malins et al. (2015), highlighting only a minor contribution to the overall lifecycle GHG emissions of transport fossil fuels, even in the highest estimates (between 0 and 2 g CO<sub>2</sub> eq/MJ;

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<sup>158</sup> For instance, a reduced crude oil processing in refineries would lead to a reduced production and availability of residual oil and petroleum coke, thereby increasing their price. This could in turn lead to a reduced consumption of such refinery outputs, or to a more likely shift to other alternative fuels that can either be “dirtier” (e.g. coal) or “cleaner” (e.g. natural gas), with all the resulting environmental implications.

Malins et al., 2015). It seems thus reasonable and acceptable to ignore emissions and impacts from military operations, whose classification as indirect effect is also not clear-cut (they do not strictly belong to other product systems, and do not take place via market-mediated mechanisms).

Changes in the quantity and type of crude oil being refined due to displacement of fossil fuels with alternative ones could result in changes to refinery co-products and their availability on the market. This may in turn lead to displacement of some of these co-products with other alternatives having different impacts during both production and use. For instance, petroleum coke might be replaced with natural gas (incurring potentially lower impacts), while fuel oil may be replaced with coal (with potentially higher impacts). However, there is currently no solid basis to draw conclusions on whether induced changes in refinery co-products is more likely to reduce or increase global impacts, or to provide any robust estimate of the possible magnitude of such effects. According to Malins et al. (2015) the climate impact from changes in refinery co-products due to fossil-fuel displacement can be in the order of  $\pm 5$  g CO<sub>2</sub> eq. per MJ of transport fossil fuel (and thus resulting in a benefit or an additional impact).

Market-mediated effects, or rebound effects, due to (marginal) changes in demand and price of petroleum products, may have significant effects on GHG emissions. However, quantification is highly uncertain and challenging, since changes and resulting impacts may occur anywhere in the global economy. Rebound effects are usually brought forward as an obstacle to the success of environmental policies, such as for renewable energy sources: the promotion of renewable energy sources creates a decrease in demand for fossil sources, which consequently leads to a decrease in their price and a rebound in their use (e.g. in cheaper -disposable-products that are consumed and wasted at higher rates). Unnasch et al. (2009) estimated that lower fossil fuel price due to lower demand incurred by displacement with alternative fuels can lead to an increase in GHG emissions from fuel supply equal to 0.25 g CO<sub>2</sub> eq./MJ. However, Malins et al. (2015) concluded that it would not be appropriate to account for price effects in life cycle impacts of petroleum-based fuel supply, due to high uncertainties and challenges to assess these effects.

As for oil spills, data from the International Tanker Owners Pollution Federation (ITOPF) on the quantities of (accidental) leakage from marine vessels used for oil transport may be used to estimate impacts due to oil spills from vessels. However, no information is available on the proportion of spills from vessels compared to overall leakage during oil extraction and transport (e.g. including also losses from pipelines). The potential climate impact of accidents (including both accidental oil spills and fires) was estimated at less than 0.01 g CO<sub>2</sub> eq./MJ of fossil transport fuel by Malins et al. (2015), although the main impact from spills is not expected to be on climate, but on affected ecosystems and landscape. Similarly, Wolf (2014) investigated, in an illustrative case study, crude oil losses and the various emissions from burning oil fields during the gulf war; the amount of crude oil burned (and related GHG emissions) were estimated at nearly 0.1% of the global crude oil consumption and of the respective emissions. Oil spills and fires are not indirect effects, but rather direct effects associated with oil supply, occurring either as a result of accidents or as a more structural property of the supply chain (e.g. structural oil losses during extraction and transport). Therefore, structural oil spills occurring during transport are normally accounted in lifecycle inventory datasets for oil supply, in terms of increased oil consumption to compensate for losses in pipelines or vessels, and related oil emissions to the environment. Potential impacts of these emissions from leakage on ecosystems and landscapes are instead typically not accounted, as not being currently captured within traditional LCA impact categories. Similarly, emissions and impacts from accidental oil spills and fires are normally not accounted in LCA, which focuses on normal (average) production conditions, disregarding accidents and risks, such as those occurring in petrochemical supply chains, including oil spills and fires. It has also to be noted

that accidents can occur also in bio-based (e.g. biofuel) supply chains, although their magnitude is likely smaller compared to oil-based chains. In light of this, Malins et al. (2015) consider that it would be appropriate to exclude emissions and impacts from accidents for both fossil-based and bio-based fuels.

Table F.1 summarises the estimated climate impact of the indirect and accident-related effects discussed above for the specific case of fossil fuels for transport (based on Malins et al., 2015), and compares it to the average total lifecycle impact of these fuels (as per Directive 2009/28/EC)<sup>159</sup>. The individual contribution of such effects is in most cases very limited (i.e. lower than 1%), while only in the case of petroleum supply protection and potential changes in refinery co-products it is or can be higher (although never larger than 6%).

**Table F.1.** Summary of the estimated climate impact of potential indirect and accident-related effects associated with the supply of fossil fuels for transport, compared to the respective average lifecycle impact (i.e. 83.8 g CO<sub>2</sub> eq/MJ fuel -Directive 2009/28/EC). Estimates are based on Malins et al. (2015).

<b>Effect</b>	<b>Estimated impact [g CO<sub>2</sub> eq/MJ of transportation fuel]</b>	<b>Contribution to total lifecycle impact (%)</b>
Road construction in occupied agricultural land (to access oil fields) and resulting iLUC	0.01 - 0.1	0.012 % - 0.119 %
Protection of petroleum supply (military operations)	0 - 2	0 % - 2.387 %
Changes in refinery co-products	± 5	± 5.967 %
Market-mediated effects	0.25	0.298 %
Accidents (oil spills and oil fires)	0.01	0.012 %

In addition to the indirect and accidents-related effects reported above, damages to landscape due to oil extraction (e.g. in the case of oil sands or accidental oil losses such as in the Niger delta) are sometimes reported as another type of indirect effect that should be taken into account when dealing with fossil-based feedstock supply. While certainly relevant, potential impacts on landscape cannot be classified as indirect effects, but rather as an additional type of impact associated with oil extraction activities. However, aesthetic/visual or ecosystem impacts due to changes in or damages to landscape are currently not captured in LCA, due to the absence of specific impact assessment methods covering these issues and related impact categories.

<sup>159</sup> Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC.

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## **Annex G - Background considerations and additional recommendations on the modelling of the use of CO<sub>2</sub> as a feedstock for plastic products**

This annex provides general methodological considerations that have been used as a background to define the rules specified in Section 4.4.5 for the modelling of the use of CO<sub>2</sub> from point emission sources as a feedstock for the production of plastic products. It also specifies additional modelling recommendations and examples, especially regarding the identification of the point of substitution when applying the Circular Footprint Formula (CFF).

### **G.1. General methodological considerations**

Most Carbon Capture and Utilisation (CCU) systems are of a multi-functional nature, i.e. the CO<sub>2</sub> source (CO<sub>2</sub>-generating process) usually provides a main product (e.g. electricity or ammonia) as well as captured or isolated CO<sub>2</sub> as co-product or by-product<sup>160</sup>. In a broader "system perspective", the whole CCU system is multi-functional, as providing both the main product from the CO<sub>2</sub> source (e.g. electricity or ammonia), as well as the final product deriving from CO<sub>2</sub> utilisation (e.g. propylene, or polyols). Moreover, the CO<sub>2</sub> utilisation (conversion) process may produce multiple products itself, thus also being multi-functional. Multi-functionality in LCA of CO<sub>2</sub> utilisation systems is an issue that has been discussed quite extensively in the literature (e.g. Giegrich et al., 2018; Zimmermann et al., 2018; Von der Assen and Bardow, 2014; Von der Assen et al, 2013). Process subdivision is the preferred way of dealing with the multi-functionality problem according to ISO 14044, however it is normally not applicable to the CO<sub>2</sub>-source (industrial plant) since CO<sub>2</sub> is produced jointly with the main product (electricity, chemicals, etc.). Other approaches including "system expansion" and allocation hence need to be applied to address multi-functionality, taking into account the goal and scope of the study (e.g. system expansion cannot be applied to perform product-specific assessments, as discussed below).

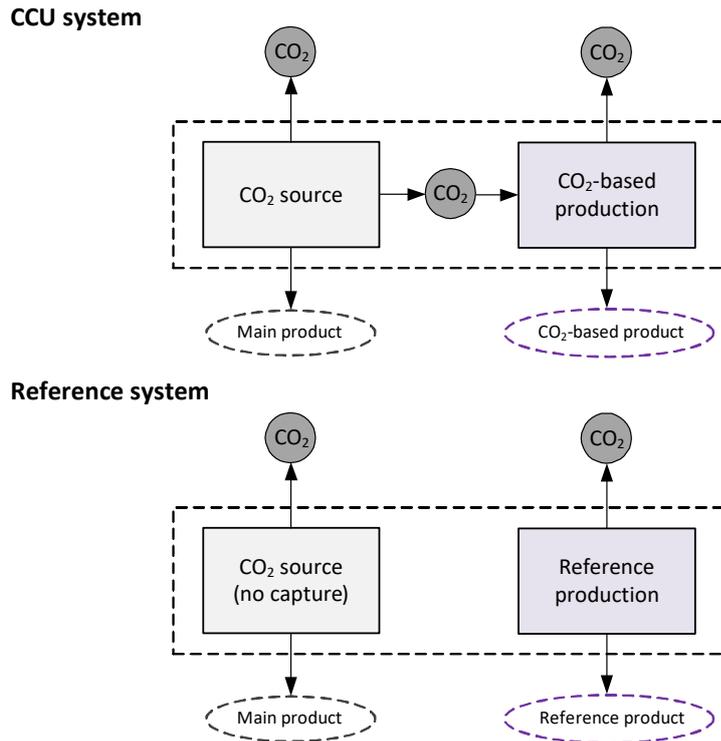
A first alternative could be to study a CCU system as a whole, considering all the co-products (or functions) it provides (e.g. electricity and CO<sub>2</sub>-based propylene, or ammonia and CO<sub>2</sub>-based polyols), according to the so-called "system expansion" approach (or anyway in an overall "system perspective"). This expanded system could then be compared with a reference system where such products are individually produced in two separate industrial systems. This approach allows to evaluate the effects of providing the products in a coupled (integrated) system (the CCU system), rather than in two separate (uncoupled) systems. Its application is hence recommended to evaluate whether establishing a CCU system for the production of polymers (or other CO<sub>2</sub>-based products) makes sense from an environmental point of view, compared to uncoupled production of the same outputs. It also allows to evaluate the potential benefits or drawbacks of process or system integration or coupling in an industrial symbiosis perspective. Figure G.1 depicts the application of the system expansion approach to a generic CCU system and the corresponding reference system including two separate production processes.

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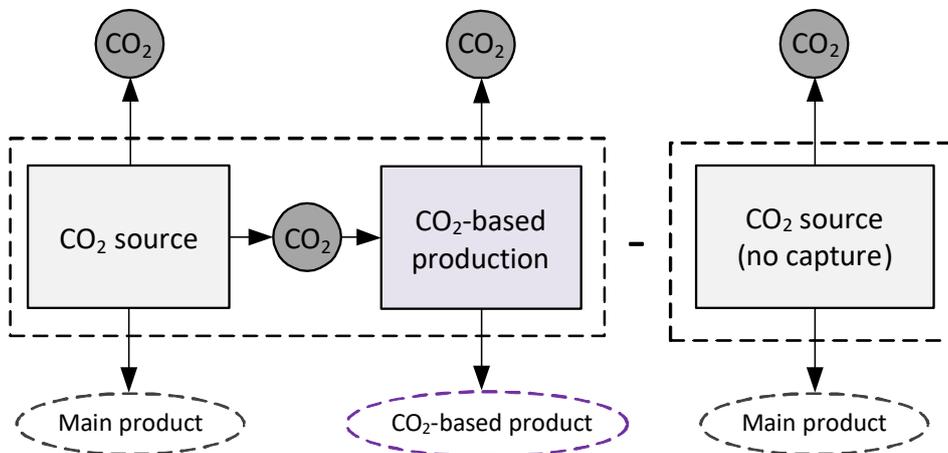
<sup>160</sup> For many emission sources (e.g. power plants, cement kilns or incineration plants) CO<sub>2</sub> capture from the generated flue gas is purposefully carried out (within the same facility) in order to allow downstream utilisation of captured CO<sub>2</sub>, which can thus be considered as the co-product from the CO<sub>2</sub> source. In other cases (e.g. ammonia production facilities), CO<sub>2</sub> separation/extraction from the main gaseous stream generated in the process (e.g. a mixture of H<sub>2</sub> and CO<sub>2</sub> in the case of ammonia production) is necessarily carried out to allow subsequent use of the other components in the process itself (e.g. the use of H<sub>2</sub> for ammonia synthesis), regardless of whether CO<sub>2</sub> is further used downstream or simply emitted to air. In this situation, the co-product of the CO<sub>2</sub> source can be considered to be the CO<sub>2</sub> stream "isolated" from the main process stream, which can then undergo additional purification steps and (similarly to CO<sub>2</sub> captured from other sources) possible compression/liquefaction for transport to downstream users.

By including all co-products and functions of the overall CCU system, the system expansion approach described above does not provide product-specific results (e.g. for the final CO<sub>2</sub>-based product of interest), which are required for assessments at the product level (as it is the case of the method presented in this document). One option to calculate product-specific results for the sole CO<sub>2</sub>-based product could be to apply system expansion via substitution (also referred to as “avoided burden approach”) to the CCU system as a whole. This approach would still be in line with the requirement from ISO 14044 to avoid allocation, and with the general hierarchy to handle multi-functionality adopted in the present method, which includes “direct substitution” as a second possible alternative (provided that a robust direct substitution can be identified; Section 4.5). The latter consists of crediting the overall (expanded) multi-functional CCU system with the production of the non-relevant co-product (i.e. the main product from the CO<sub>2</sub> source such as electricity or ammonia) as carried out in an uncoupled system without CO<sub>2</sub> capture (Figure G.2). For instance, in the case of combined ammonia and polyol production in a coupled CCU system, substitution would imply subtracting the burdens from the production of sole ammonia in an uncoupled system without CO<sub>2</sub> capture. Similarly, in the case of CO<sub>2</sub> capture from a power plant, the process of electricity generation without CO<sub>2</sub> capture would be credited to the overall, expanded CCU system providing both electricity and the CO<sub>2</sub>-based product. However, applying this approach may result in a product system having negative emissions, which may be misleading in that the system seems beneficial to the environment (uptake of emissions). Moreover, an uncoupled “mono-functional” production process to be considered for substitution may not be available for all CO<sub>2</sub> sources, while if more processes exist the selection is not straightforward (Von der Assen et al., 2013).

Direct substitution may also be applied to address multi-functionality of the CO<sub>2</sub> source (e.g. a power plant delivering both electricity and captured CO<sub>2</sub>) within a specific CO<sub>2</sub>-based supply chain, rather than at the level of a CCU system as a whole (as discussed above). In this case, the CO<sub>2</sub> source (and not the CCU system) would be credited with the production of the non-relevant co-product (e.g. electricity or ammonia) as carried out in a process without CO<sub>2</sub> capture. However, considering the typical main product of a CO<sub>2</sub> source (e.g. electricity) as the co-product for substitution, and captured/isolated CO<sub>2</sub> as the relevant co-product, can be deemed unrealistic under current production conditions, which can lead to (further) misinterpretations. The use of system expansion via substitution as the only way of dealing with multi-functionality of CCU systems and of specific CO<sub>2</sub> sources is therefore discouraged (also considering the issues discussed above).



**Figure G.1.** System expansion approach to compare an overall CCU system with a conventional (reference) system considering uncoupled production of the same product outputs. The main product of the CO<sub>2</sub> source (with capture) is included in the functional unit and the reference system is expanded to include the production of the main product in a process without CO<sub>2</sub> capture. Adapted from Zimmermann et al. (2018).



**Figure G.2.** Application of system expansion via substitution to a CCU system to quantify the burdens associated with production of the CO<sub>2</sub>-based product alone. The overall CCU system is expanded to include avoided production of the main product of the CO<sub>2</sub> source in a process without CO<sub>2</sub> capture.

An alternative to system expansion via substitution is the allocation of environmental burdens between co-products based on a common underlying relationship. Similarly to direct substitution, allocation could be either applied at the overall CCU system level with its co-products (e.g. ammonia and polyols, or electricity and methanol) or at the level of CO<sub>2</sub> source as the actual multi-functional process in a specific CO<sub>2</sub>-based supply chain. However, allocation is normally not applied at a system level in common LCA practice, but to specific multi-functional processes within the life cycle of a product, such as the CO<sub>2</sub>-generating process

(CO<sub>2</sub> source) for a specific CO<sub>2</sub>-based product. In this case, allocation would need to be applied between the main product of the CO<sub>2</sub> source (e.g. electricity or ammonia) and captured/isolated CO<sub>2</sub> before any further processing and transport for subsequent use<sup>161,162</sup>. As for direct substitution discussed above, this approach inherently implies considering that captured/isolated CO<sub>2</sub> is a co-product (or by-product) of the CO<sub>2</sub> source, which is not straightforward to establish (as briefly discussed below, and more widely in Giegrich et al., 2018). However, if a demand exist for captured CO<sub>2</sub> (as it is the case once a CCU system is in place) it can be reasonably considered a co-product, and the following considerations on how allocation could be performed refer to this specific situation.

According to ISO 14044, a common underlying physical relationship among co-products should be applied as a first option for allocation, such as mass or energy relationships. Only when a common physical relationship cannot be identified, such as between captured CO<sub>2</sub> and electricity from a fossil-fired power plant or between electricity and methanol from an overall CCU system, another appropriate relationship among co-products should be identified and applied, such as their economic value. For many CO<sub>2</sub> sources (e.g. power plants or waste incineration plants), no common physical relationships exist among co-products (e.g. energy and captured CO<sub>2</sub>), and the same applies to the overall CCU systems relying on such sources (providing e.g. electricity and methanol). Similarly, no physical relationships that properly reflect the relative inherent value of such co-products can be frequently identified (e.g. ammonia and isolated CO<sub>2</sub> from the same process currently have a very different economic value). Therefore, physical allocation is usually not appropriate for those sources and systems, while economic allocation appears a better alternative to reflect differences in the value of co-products. However, due to the innovative nature of CCU pathways and systems, and hence to the absence of an established market for at least some of the respective co-product outputs, choosing an appropriate economic value for the different co-products may prove challenging (especially if allocation is to be performed at the level of the CO<sub>2</sub> source between captured/isolated CO<sub>2</sub> and the main co-product(s)). Additionally, the economic value of the main product and co-products may change over time and may differ from one geographical location to another, which adds further uncertainties. An alternative may be to rely on production costs of each co-product, but information on these are difficult to derive for captured/isolated CO<sub>2</sub> (and are potentially equal to zero for several sources). For these reasons, the use of economic allocation cannot be currently recommended.

All the alternatives discussed above to deal with multi-functionality of CO<sub>2</sub> sources (i.e. direct substitution and allocation) are based on the assumption that captured or isolated CO<sub>2</sub> is a co-product of the CO<sub>2</sub> source itself. However, in the current situation of large availability of raw gaseous CO<sub>2</sub> for possible capture for many sources, and in the absence of an established market and demand for captured CO<sub>2</sub>, it is not straightforward to establish whether captured/isolated CO<sub>2</sub> should be considered a co-product of the CO<sub>2</sub> source, or if raw gaseous CO<sub>2</sub> should be rather considered a waste from the same source. According to Giegrich et al. (2018), raw gaseous CO<sub>2</sub> can be considered as a waste if it has no economic value before capturing, or if its supply/availability is much larger than its demand in the context of the specific CO<sub>2</sub> emitters (power plants, cement kilns, etc.).

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<sup>161</sup> Indeed, the burdens of any purification and compression/liquefaction step possibly performed after raw CO<sub>2</sub> is captured or isolated from the gaseous stream generated from the considered source can generally be distinguished and attributed to captured CO<sub>2</sub>, and hence to the relevant CO<sub>2</sub>-based product. No allocation is thus needed in this case, although this ultimately depends on the level of disaggregation/granularity of available process data for the CO<sub>2</sub> source. Similar considerations apply also to any subsequent transport of captured CO<sub>2</sub> to downstream users.

<sup>162</sup> The utilisation process receiving captured CO<sub>2</sub> as an input may also be multi-functional (i.e. it may provide multiple product outputs). However, in this case, the general rules to handle multi-functionality situations outlined in Section 4.5 regularly apply.

When raw gaseous CO<sub>2</sub> generated by the source is considered a waste, the question whether it is a waste for disposal or for recycling/recovery may arise. In case of recycling, the issue of which product systems should carry the burdens of common processes across different life cycles (e.g. supply of carbon-providing feedstock and final disposal of the CO<sub>2</sub>-transferred to the CO<sub>2</sub>-based product) also needs to be addressed. However, the option of raw CO<sub>2</sub> being a waste for disposal is automatically excluded by its subsequent utilisation, which inherently implies that some form of recycling/recovery is undertaken. Based on current availability and demand (and absence of an economic value before capturing), at present raw gaseous CO<sub>2</sub> could thus be considered as a waste for recycling, which is supplied in much larger quantities than its current demand for subsequent capture and utilisation. However, this assumption may need to be reconsidered in a possible future situation of lower availability of raw CO<sub>2</sub> from specific emission sources (thanks, for instance, to energy decarbonisation) and wider use of CO<sub>2</sub> as a valuable resource for a broader range of applications (although being thermodynamically not advantageous).

## **G.2. Additional modelling recommendations (identification of the point of substitution for CFF application)**

According with the modelling requirements specified in Section 4.4.5, the Circular Footprint Formula shall be applied to model the use of CO<sub>2</sub> from point emission sources as a feedstock for plastic products. This requires prior identification of the most proximate and suitable point of substitution within the supply chain, i.e. of the closest point with respect to CO<sub>2</sub> generation where a “recycled” product from the CO<sub>2</sub>-based pathway replaces an equivalent (conventional) product normally produced from primary (e.g. fossil-based) resources. In this respect, the following considerations are made.

The point of substitution could be in principle identified in correspondence of the input to the first CO<sub>2</sub> utilisation process, with captured CO<sub>2</sub> (which may be considered to represent the very first “recycled” product of the CO<sub>2</sub>-based pathway) directly replacing the use of a specific substance or product regularly produced from primary resources in the same process. For instance, in the case of CO<sub>2</sub>-based *polyols*, captured CO<sub>2</sub> may be assumed to totally or partially replace fossil-based *propylene oxide* conventionally used, along with other raw materials, for polyols production. Similarly, in the case of CO<sub>2</sub>-based *methanol* used as a building block for CO<sub>2</sub>-based olefins (*ethylene* and *propylene*) production, captured CO<sub>2</sub> may be considered to replace *carbon monoxide (CO)* from natural gas reforming as an input to *methanol* production. However, CO<sub>2</sub>-based synthesis pathways normally differ at least partially from conventional (fossil-based) pathways in terms of: (i) material and energy inputs and outputs (as in the case of e.g. CO<sub>2</sub>-based *polyols*), or (ii) of applied conversion routes/technology (as in the case of CO<sub>2</sub>-based *methanol*, which is mostly derived from direct CO<sub>2</sub> hydrogenation rather than the complex of reactions involved when CO/CO<sub>2</sub>-containing synthesis gas is used as an input). In some cases, differences exist also in further downstream conversion/synthesis processes, such as in the case of CO<sub>2</sub>-based olefins, whose alternative synthesis from (CO<sub>2</sub>-based) methanol completely differs from the conventional synthesis route relying on cracking of naphtha and/or other hydrocarbons.

Based on the considerations and examples above, it is hence deemed more appropriate to consider the point of substitution in correspondence of the input to the first identical conversion/synthesis process within both the CO<sub>2</sub>-based supply chain and the replaced conventional pathway<sup>163</sup>. This would mean considering that the CO<sub>2</sub>-based “recycling chain” extends until the process delivering the first useful

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<sup>163</sup> A similar situation applies, for instance, to aluminium recycling, where secondary aluminium production is based on a process having a different energy consumption compared to primary aluminium. The point of substitution is hence frequently identified at the level of finished secondary aluminium ingots, replacing primary ingots as an input to manufacturing processes.

CO<sub>2</sub>-based product that actually replaces an equivalent conventional product from primary resources as input to a same identical conversion/synthesis process in the supply chain. In this perspective, which should be adopted in the modelling, CO<sub>2</sub>-based *polyols* may be assumed to replace fossil-based ones, while CO<sub>2</sub>-based *propylene* from the methanol-to-olefin route may be considered to replace conventional *propylene* from naphtha cracking. If the process directly using captured CO<sub>2</sub> as an input is identical to the one relying on the replaced conventional product, the point of substitution may still be set at the level of captured CO<sub>2</sub> ready for use in such process.

### G.3. References

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## **Annex H - General considerations on the assessment of products based on maturing or emerging technologies**

A popular formulation of the progress principle states that “the cost input per unit (of product) declines at a uniform rate with cumulative production” (Dutton and Thomas, 1984). Moreover, it is by now proven that companies and industries generally go through a learning curve, meaning that their efficiency and productivity increase as their experience (i.e. cumulative production) increases. While the shape of the learning curves varies depending on several parameters, the rate of improvement appears often to be higher at the beginning of production (i.e. progress is faster) than for mature industries. For instance, Dutton and Thomas (1984) analysed data from many different industries and found a distribution of progress ratios with a mode between 81% and 82%. This implies that for every doubling in cumulative output, unit costs decrease to 81% or 82% of their former value, or also that the learning rate of the technology is equal to 18-19%. These values have been further substantiated; for instance, Junginger et al. (2010) found a range of progress ratios similarly centred around 83-85% for various energy technologies, but with strong variations across the spectrum of technologies analysed.

The learning rates depend on several factors, including (Figure H.1): technological improvements, organizational practices, organizational characteristics, and the type of learning in which an organization engages (EPA, 2016).

Expanding further on this concept is beyond the scope of this document, however, it is important to acknowledge the importance of learning curves when assessing products relying on technologies or processes at different levels of development/maturity and/or running at different scales of production (e.g. Polylactic Acid -PLA- production). This is especially relevant when considering at the same time products based on emerging technologies at an early stage of development (e.g. production of Hydroxymethylfurfural<sup>164</sup>), and products relying on more established/mature technologies running at larger scales (e.g. Polyethylene Terephthalate – PET).

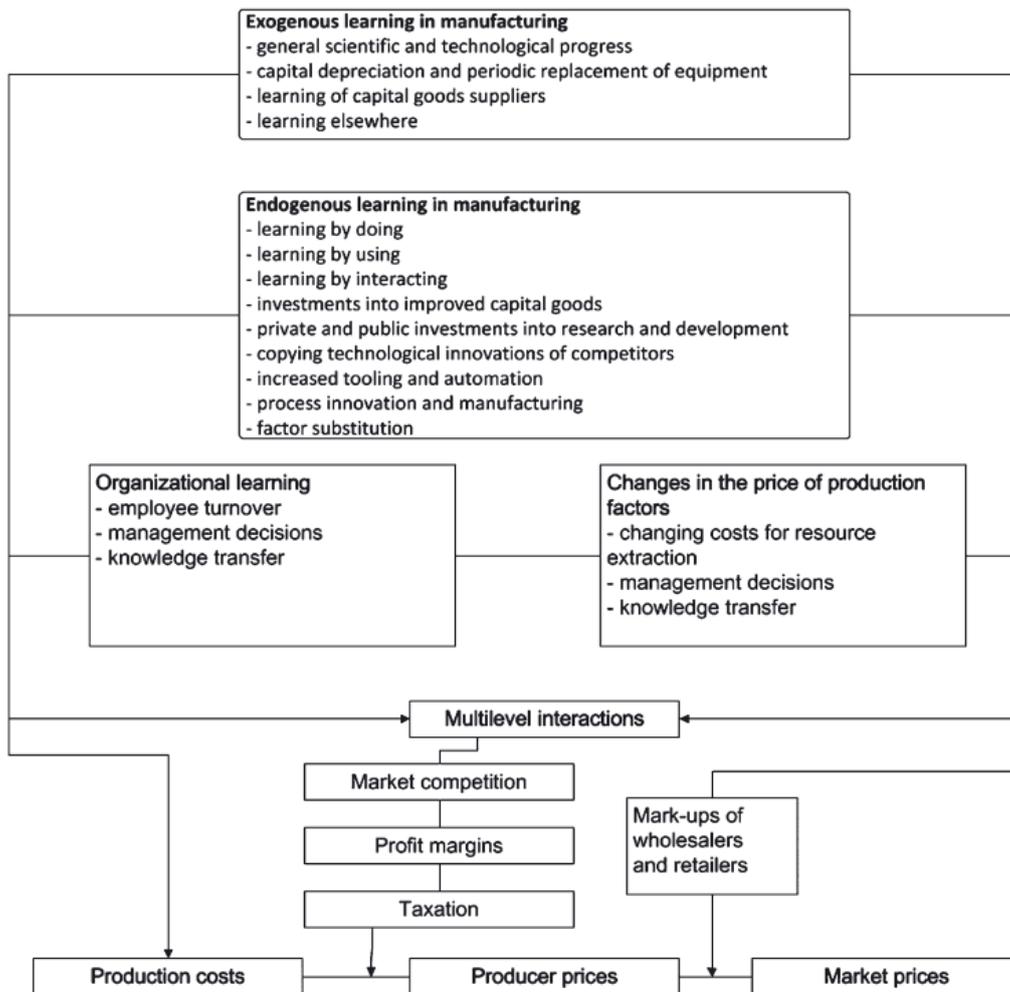
For instance, over the last 40-50 years, production of Purified Terephthalic Acid (PTA, a precursor of PET) has benefited from improvements in chemical conversion yields (up to 96%), energy efficiency across the process, solvent consumption, purity of the output, as well as in the valorisation of by-products and recycling rates of catalyst materials<sup>165</sup>. Overall, it is thus safe to assume that the environmental impacts of PTA have decreased with the cumulative production of the polymer.

While broad ranges of learning rates can be assigned to different industries, these values are subject to wide volatility and uncertainty. For instance, far from being constant values, learning rates can also be subject to discontinuities and strongly influenced by knowledge depreciation and knowledge forgetting (EPA, 2016). For these reasons, it is often difficult to predict future learning rates (Daugaard, 2015), or extrapolate existing rates to newer products or industries.

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<sup>164</sup> Hydroxymethylfurfural is a precursor to PEF (Polyethylene Furanoate), a polymer obtained from polymerisation of Furandicarboxylic Acid (FDCA) and Mono-Ethylene Glycol (MEG).

<sup>165</sup> Based on direct information provided by one stakeholder during the technical consultation processes conducted during the development of this document.



**Figure H.1.** Exogenous and endogenous factors that affect learning rates in the manufacturing industry (Weiss et al., 2010).

It is acknowledged that production of polymers from alternative feedstock sources relies in several cases on processes which are at the on-set or in the middle of their learning curves, and that may undergo (important) optimisation, further development, and scale effects in the future. If this will actually occur, the environmental profile of products based on such polymers will likely improve.

For products relying on emerging (early-stage) technologies (e.g. products based on Polyethylene Furanoate or CO<sub>2</sub>-based polymers), Life Cycle Inventory data from real (full-scale) production plants are normally not available, although at least partial data from real facilities may be available to companies running pilot or demonstrative plants based on such technologies. In some cases, data from theoretical process simulation are available or can be produced, especially for chemical synthesis processes, and generally referring to production capacities comparable to those of pilot plants (in the order of thousands tons per year). While these data are normally generated taking into account some process optimisation strategies (e.g. energy integration), they usually do not account for possible (efficiency) improvements due to upscaling, further process integration, experience and/or other process optimisation strategies. Moreover, they may also provide only a partial coverage of the environmental burdens of the process (e.g. waste flows and certain direct emissions may be excluded from the simulation), although energy and material flows that are most relevant in a LCA context are generally covered. On the other hand, simulation parameters may be set so as to reproduce

(extremely) optimised process conditions that are not (yet) reflected in reality, which should be avoided. Therefore, when data from process simulation are used in the LCA study of a specific product, a fully consistent and reliable comparison with products relying on (more) mature technologies cannot be generally accomplished. If such an evaluation is conducted (limited to company-internal studies if the *Plastics LCA* method is followed, being the product in most cases non-commercially available), limitations related to applied data shall be clearly acknowledged in the study, and taken into account in the interpretation of the LCA results, along with differences in maturity/development level of technologies used in the supply chain of compared products.

In some cases, emerging conversion/production pathways are similar to those applied in more established processes. In this case, available data related to such more established process may be applied as a proxy in the modelling of the emerging process of interest. While this has the advantage to inherently account for the effects of process optimisation, efficiency improvement, and scale effects, the applied data may not be sufficiently representative of the process to be modelled, which shall be properly taken into account in the interpretation of the LCA study results.

For products relying on more established but still evolving or maturing production technologies (e.g. Bio-PBS<sup>166</sup>, PLA<sup>167</sup>, PLA-based and starch-based products), Life Cycle Inventory data are generally available. However, these normally refer to processes running at lower scales compared to products based on more established technologies, and still showing a potential for further optimisation and (efficiency) improvement (with a subsequent possible reduction of the respective potential impacts). In this case, possible “scaling factors” (or learning rates) reflecting potential future improvements may be ideally applied, based, for instance, on historical improvement experienced in similar, more established conversion processes (e.g. those used in conventional polymer production), provided that specific input(s) or output(s) potentially affected by such improvement is known.

However, reliable data on improvement experienced by specific processes are hardly available, while applying any generic learning rate from the literature to specific processes is likely to introduce substantial additional uncertainty, as discussed above. Moreover, any improvement rates cannot be applied when production inventories are only available in an aggregated form (as it is currently the case of many bio-based polymers), as no changes can be made to specific processes quantities, elementary flows, or other relevant parameters affected by the expected improvement.

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<sup>166</sup> Bio-based Polybutylene Succinate.

<sup>167</sup> Polylactic Acid.

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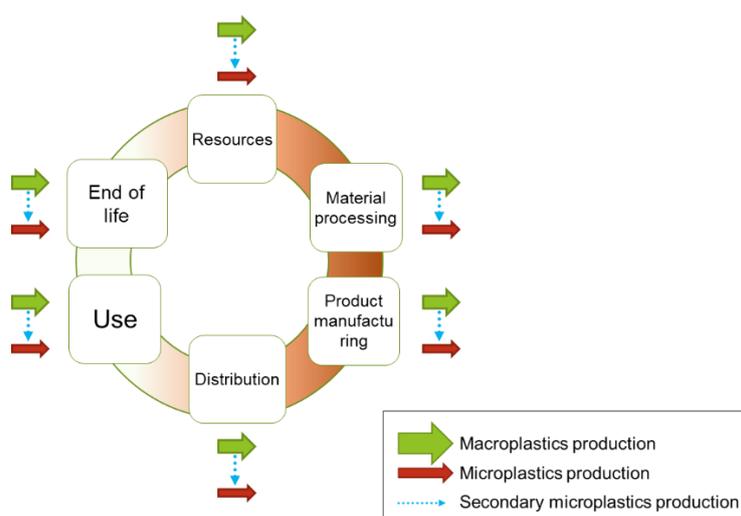
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## Annex I - Methods and approaches to quantify macro-and micro-plastics generation and release throughout the product life cycle (including product litter)

This Annex provides a brief overview of the main approaches and methods currently available to quantify potential generation and release of macro-plastics and (secondary) micro-plastics in the open environment from different sources along the life cycle of a product. It also provides an operational description of the *Plastic Leak Project* (PLP) method (Peano et al., 2020), which may be used to calculate additional environmental information related to the contribution of the product in scope to generation and release of macro- and micro-plastics in the terrestrial, freshwater and/or marine environment (ocean) throughout the life cycle. As reported in Section 3.2.5, this additional information should be provided in the LCA study.

### I.1. Introduction

Plastic littering and debris have become of increasing concern due to their potential impacts on ecosystems, landscape and, indirectly, human health (Thevenon et al., 2014; Arroyo Schnell et al. 2017). Macro-plastics can be littered along the life cycle of products due to either direct actions from consumers (e.g., beverage bottles directly discarded on beaches) or indirect mechanisms (e.g., indirect release from mismanaged landfill sites or dumpsites), leading to emissions to the terrestrial, freshwater and/or marine environment. Moreover, released macro-plastics can eventually transform into secondary micro-plastics after fragmentation and degradation in the environment (Figure I.1). Both macro- and micro-plastics can negatively impact (marine) ecosystems, including biodiversity, for instance directly via entanglement, suffocation or ingestion by different animal species, or indirectly via transport and introduction of non-indigenous alien species (e.g. Deudero and Alomar, 2015). They can also have an indirect impact on human health through diets, due for instance to concentration of toxic substances present in polluted (sea)-water on the surface of floating plastic particles, and subsequent accumulation in the food chain (e.g., micro-plastics and any adsorbed pollutant may end up in marine salt used for human consumption) (Yang et al., 2015). This has emphasised the importance of working towards the integration of these impacts in LCA studies, as specified for instance in the “Medellin Declaration” with a specific focus on marine litter (Sonnemann and Valdivia, 2017).



**Figure I.1.** Generation of macro- and secondary micro-plastics along the life cycle of products.

## I.2. Overview of approaches and methods to quantify macro- and micro-plastics emissions and impacts in LCA

One of the most comprehensive approaches currently available to quantify macro- and micro-plastics emissions throughout the product life cycle, based on state-of-the-art knowledge and data, is the one developed within the *Plastic Leak Project*<sup>168</sup> (PLP) (Peano et al., 2020), which is here referred to as the *PLP method*. The *PLP method* has been developed and validated based on a multi-stakeholder initiative including 35 public, private and scientific organizations, with the aim of providing companies with scientific-based guidelines to map, measure and forecast plastic leakage along the value chain. The PLP guidelines specify calculation approaches and default data applicable to estimate the loss and release of macro- and micro-plastics to the environment from different relevant sources throughout the life cycle, and considering different end environmental compartments (i.e. terrestrial, freshwater and marine environment) and redistribution among initial compartments. However, not all the commonly identified sources of micro-plastics have been included in the *PLP method* (e.g. marine coatings and road markings are excluded), due to the lack of sufficiently reliable, accurate and/or agreed data or calculation methods for quantification<sup>169</sup>. Moreover, the possible fate of released (macro)-plastics into the environment is not addressed (e.g. fragmentation or degradation). Further details on the main features of the *PLP method* are provided in Section I.3, including an operational description of the main equations and parameters applicable to quantify generation (“loss”) and release of macro- and micro-plastics along the product life cycle.

In a somehow parallel initiative to the *PLP* project, Ciroth and Kouame (2019) elaborated a framework to estimate plastic (litter) generation (total loss of plastic parts)<sup>170</sup> associated with background datasets of the *ecoinvent* database<sup>171</sup> (v. 3.5; cut-off system model; Wernet et al., 2016), by means of plastic littering probabilities calculated based on available literature<sup>172</sup>, and using the *openLCA* software<sup>173</sup>. The framework is based on the probability of each flow and process included in a dataset (i.e. of each inventory exchange) to take part in a littering event. For this purpose, flows and processes of *ecoinvent* 3.5 datasets are classified according to a risk for littering (“littering event probability”) for each exchange. The littering event probability is classified for each flow/process in six categories, from “none” (zero probability) to “very high” (0.95 probability). Moreover, littering event probabilities for processes consider also the type of system where the process is taking place (i.e. closed or open system). Once flows and processes are characterized according to the respective littering probability, the risk for litter is aggregated for each process dataset, yielding an emission of “plastic parts, small” for each dataset (i.e. process). Plastic litter is thus characterised only as “plastic parts, small”, without specification of the plastic size, shape, emission compartment, nor plastic type. The approach only quantifies the total loss of plastic (“plastic parts”) for a specific inventory dataset (i.e. from a process/activity) without considering the actual initial emission to a specific compartment, potential transfer among different compartments or reintroduction into the technosphere

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<sup>168</sup> <https://quantis-intl.com/metrics/initiatives/plastic-leak-project/>

<sup>169</sup> Note that, however, these sources are addressed in Boucher & Friot (2017), who initially developed some of the quantification approaches lately implemented (with possible adaptations) in the *PLP method*.

<sup>170</sup> Generated estimates can be interpreted as plastic “losses” from processes (following the terminology used in the *PLP method*) as they do not specify the emission compartment nor account for potential transfer among compartments or possible reintroduction into the technosphere (e.g. thanks to collection or capture systems).

<sup>171</sup> <https://nexus.openlca.org/database/ecoinvent>

<sup>172</sup> Multiple data sources were considered for defining the littering probabilities, including Astrup et al., (2009); Sundt et al. (2014); Essel et al. (2015); Jambeck et al. (2015); Lassen et al. (2015); Cole and Sherrington (2016); Magnusson et al. (2016); Geyer et al. (2017); Kawecki et al., (2018); and Ryberg et al. (2019).

<sup>173</sup> <http://www.openlca.org/>

(e.g. through collection or capture systems), nor the fate of the released plastic once in the environment (e.g. fragmentation, degradation).

No other structured approaches or methods are currently available to quantify macro- and/or micro- plastics generation/release into the environment in a life cycle perspective. However, there are some studies that provide potentially useful data that may be applied for this purpose. For instance, the supporting study to the EU Directive 2019/904 on Single Use Plastics (SUPs) (European Union, 2019) provides estimates of the littering rate of some SUP items (ICF and Eunomia, 2018), which can be useful to estimate the total release of such products as macro-plastics into the environment (although without distinguishing among compartments), and to define average End of Life scenarios accounting for product littering. Moreover, harmonised data on observed beach litter from surveys conducted across different EU Member States during the period 2012-2016 are available in Addamo et al. (2018) and Hanke et al. (2019), covering also specific plastic products. These data may be used to estimate the share of a product ending up as marine litter (i.e. product-specific marine litter rates), based on product consumption statistics over an appropriate time period. However, such estimates would be affected by even considerable uncertainty, due to, for instance, representativeness of collected beach litter data (e.g. in terms of surveyed coastline length and respective geographical distribution across EU), discrepancies between amounts observed on beaches and real presence in the marine environment (surface, water column and sediments), and absence of actual modelling of transport and fate of the product in the environment starting from the respective source.

As for the evaluation of the potential impacts of macro- and micro-plastics ultimately released (e.g. littered) into the environment, no sufficiently complete and robust impact assessment models are currently available. This is mainly due to still incomplete understanding of the underlying mechanisms governing the full fate, exposure, and subsequent (physical and toxicological) effects on ecosystems and humans of plastic products and particles released into the environment, which makes the development of a suitable impact assessment method challenging. For instance, regarding fate, while some information on degradation and bio-degradation is available for some plastic polymers under specific environmental conditions or in specific compartments, data do not cover all types of polymers in a comprehensive manner. Moreover, in general, degradation and bio-degradation pathways of plastic products once they have reached a specific end compartment (and possibly during transport from one compartment to another) still have to be further investigated and understood, especially in terms of generated (bio)-degradation products ultimately released in the environment (e.g. additives, other degradation compounds, micro- and nano-plastics, etc.). However, the ongoing MarILCA project<sup>174</sup> is working on the development of a framework to integrate the potential environmental impacts of marine litter in LCA, which may provide sufficiently robust, complete and agreed impact assessment methods and indicators to quantitatively address the potential impacts of at least those plastic products and particles ultimately ending up in the marine environment. Other approaches have also been recently proposed to individually evaluate specific impacts from macro- or micro-plastics released into the environment. For instance, Woods et al. (2019) proposed an approach to quantify the impacts from entanglement on marine species (i.e. marine biodiversity) due to macro-plastic debris, focusing on the calculation of a preliminary effect factor (i.e. without considering the full fate, exposure and effect mechanism governing such impact). On the other hand, Saling et al. (2020) developed and tested an impact assessment model to quantify ecotoxicological impacts of micro-plastics released in the marine environment,

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<sup>174</sup> The MarILCA project (<https://marilca.org/>) is a joint effort of the UN Life Cycle Initiative and of the Forum for Sustainability through Life Cycle Innovation (FSLCI).

considering both the fate of micro-plastics and their specific eco-toxic effects towards different organisms.

### **I.3. The *PLP method*: operational description**

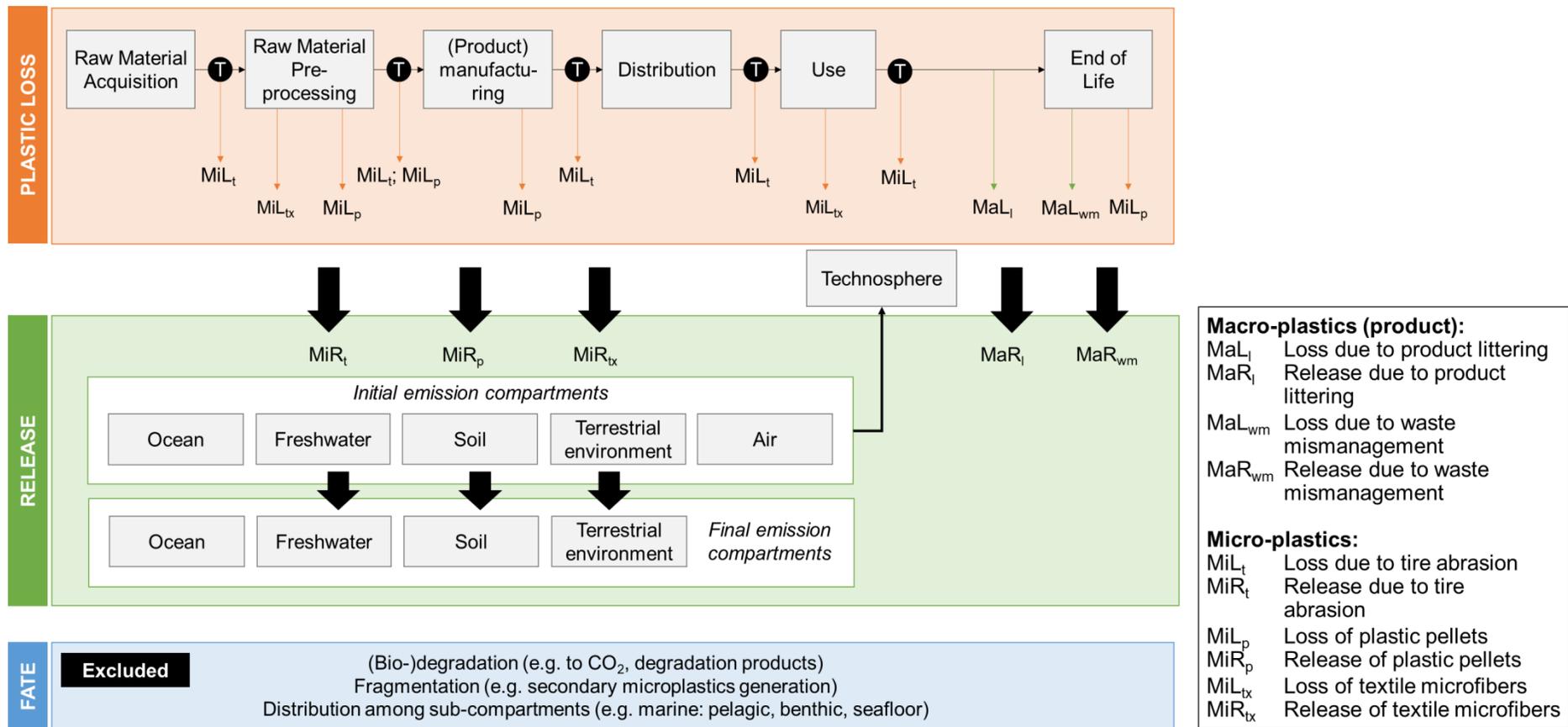
This section describes the main general features of the *PLP method* and the related calculation procedures (including relevant parameters and default values) which may be applied to quantify potential generation and release of macro- and micro-plastics throughout the product life cycle for the purpose of providing additional environmental information on the product in scope. A complete description of the method and of the respective parameters and limitations is available in the official Plastic Leak Project report (Peano et al., 2020). Note that the terminology used in the following description may have been partially changed compared to the original one, without altering the meaning.

#### **I.3.1. General characteristics**

The *PLP method* addresses emissions (release; "R") of plastic products as macro-plastics at the End of Life stage, due to product littering by consumers (MaR<sub>i</sub>) and waste mismanagement (MaR<sub>wm</sub>), such as from poorly managed landfill sites or dumpsites (e.g. in developing countries). It also addresses emissions of micro-plastics from tire abrasion (MiR<sub>t</sub>), plastic pellets (MiR<sub>p</sub>) and textile fibres (MiR<sub>tx</sub>), from relevant supply-chain stages (Table I.1). The method differentiates between plastic loss ("L") from the "technosphere" (i.e. the amount of macro- or micro-plastics directly generated from processes or consumers) and their actual final release to the environment (terrestrial, freshwater and marine). Final release accounts for possible take-back from the technosphere (e.g. through dedicated capture systems or littering collection) and for the redistribution of released plastics among the different environmental compartments (Figure I.2). While default loss, release and redistribution rates specified in the *PLP method* reflect state-of-the art knowledge and data on related relevant aspects, in some cases uncertainty and knowledge gaps may still be significant. The user of this method is thus encouraged to check for availability of updated, less uncertain and/or more representative data, and to apply them in the LCA study, provided that they are adequately documented and justified.

**Table I.1.** Sources of macro- and micro-plastics loss (L) and emissions (release, R) considered in the *PLP method* and life cycle stages where these are expected to occur (all commonly identified micro-plastics sources are reported for completeness, based on Boucher and Friot, 2017).

Sources of macro- and micro-plastics loss and release	Life cycle stage(s)	Addressed in the <i>PLP method</i>
<i>Product littering</i> (macro-plastics) <b><i>MaL<sub>l</sub> ; MaR<sub>l</sub></i></b>	End of Life	X
<i>Waste mismanagement</i> (macro-plastics) <b><i>MaL<sub>wm</sub> ; MaR<sub>wm</sub></i></b>	End of Life	X
<i>Plastic pellets</i> (micro-plastics) <b><i>MiL<sub>p</sub> ; MiR<sub>p</sub></i></b>	Raw Material Pre-processing (pellet production), Manufacturing, Distribution and transport, End of Life	X
<i>Tyre abrasion</i> (micro-plastics) <b><i>MiL<sub>t</sub> ; MiR<sub>t</sub></i></b>	Distribution and road transport in general	X
<i>Synthetic textiles</i> (micro-plastics) <b><i>MiL<sub>tx</sub> ; MiR<sub>tx</sub></i></b>	Raw Material Pre-processing (textile production), Use stage (washing)	X
<i>Marine coatings</i> (micro-plastics) <b><i>MiL<sub>mc</sub> ; MiR<sub>mc</sub></i></b>	Distribution and water transport in general	Not considered
<i>Road markings</i> (micro-plastics) <b><i>MiL<sub>rm</sub> ; MiR<sub>rm</sub></i></b>	Distribution and road transport in general	Not considered
<i>Personal care products</i>	Use stage	Not considered
<i>City dust</i>	-	Not considered



Note: Final release of macro-plastics occurs only to ocean and terrestrial environment. Releases to ocean and freshwater include both plastics/particles deposited in sediments and those suspended in the water column.

**Figure I.2.** Macro-plastics (Ma) and micro-plastics (Mi) losses (L) and releases (R) considered in the *PLP method*: sources, mechanisms and final emission compartments.

### I.3.2. Macro-plastics loss and release due to product littering and waste mismanagement at End of Life

The total mass of plastic waste that is considered to be lost as macro-plastics at the End of Life of a specific plastic product (indicated as Mass of Plastic Waste loss;  $MPW_i$ ) is calculated as the share of product waste subject to littering events and waste mismanagement practices (e.g. direct discharge of waste to waterways, uncollected waste, poor management, etc.), according to Equation I.1 (where " $m_{prod,EoL}$ " is the total mass of product leaving the Use stage and entering End of Life, per functional unit). Default values of littering rates ( $LR_{lit}$ ) are estimated based on product size –i.e. *small or detachable (<5cm)*, *medium size (5-25 cm)* and *large size (>25cm)* – and use –i.e. *in-house (non-flushable)*, *in-house (flushable)* and *on-the-go-*, and range from 0% to 5% (Peano et al., 2020; p. 75). The share of mismanaged waste (MWI) is a country-specific parameter, calculated by default based on World Bank data related to national waste management practices (Kaza et al., 2018) (as reported in the *Plastic Leak Project Sectorial Guidances Generic Data v1.0*). For application to an EU average context, a weighted average of mismanaged waste according to the population of each EU country may be calculated (see Table I.2). However, country-specific values for individual Member States should be applied, where relevant, depending on the geographical scope of the LCA study. Alternative values may be applied in the LCA study for the littering rate and the share of mismanaged waste, provided that they are adequately documented and justified.

$$MPW_i(kg) = (MaL_l + MaL_{wm})(kg) = m_{prod,EoL}(kg) * (LR_{lit}(\%) + (1 - LR_{lit}(\%)) * MWI(\%))$$

**Equation I.1** – Macro-plastics loss due to product littering and waste mismanagement at End of Life (see Table I.2 for the definition of parameters).

Not the entire mass of plastic waste lost from the technosphere ( $MPW_i$ ) is considered to be eventually released to the environment, since (littered) plastic waste has a residual value leading to further potential collection through informal systems (e.g. waste pickers). Following expert judgment and interviews, default values of initial release rates to ocean ( $RelR_{ocean}$ ), freshwater ( $RelR_{frw}$ ), and terrestrial environment ( $RelR_{terenv}$ ) are defined based on the size of the product<sup>175</sup> –i.e. *small (<5cm)*, *medium (5-25 cm)* and *large (>25cm)*– and the residual value of the waste material –i.e. *low*, *medium* or *high* (Peano et al., 2020; p. 79). Default initial release rates range from 1% to 95%, with the remaining share of the product after release that is considered to be informally collected. No differentiation is explicitly made between (plastic) products made of different materials (polymers), although the type of polymer inherently affects the proposed residual value; polymers with higher domestic recycling rate on a given market (and thus higher residual value) have a larger probability to be picked by scavengers and less chance of being released into the environment.

The actual macro-plastics release ( $MaR$ ) to final environmental compartments (i.e. ocean and terrestrial environment) is finally calculated based on redistribution rates ( $RedR$ ), which are used as an approximation to model the partial fate of initially released plastic waste, considering its possible redistribution among different environmental compartments. As default, all plastic waste initially released to freshwater is assumed to eventually reach the ocean, while plastic waste initially released to terrestrial environment and ocean is not transferred to other compartments (i.e. a redistribution rate of 100% is assumed in both cases; Table I.2). The calculation is conducted according to Equation I.2 (see Table I.2 for the definition of parameters). Alternative values may be applied in the LCA study for both release and redistribution rates, provided that they are adequately documented and justified.

<sup>175</sup> Inherently accounting for the product shape, with smaller objects expected (and considered) to be more easily transported by wind or water flows.

$$MaR_{ocean} = MPW_l(kg) * (RelR_{ocean} (%) + RelR_{frw} (%)) * RedR (%) \quad (a)$$

$$MaR_{terenv} = MPW_l(kg) * RelR_{terenv} (%) * RedR (%) \quad (b)$$

**Equation I.2** – Macro-plastics release to ocean (a) and terrestrial environment (b) due to product littering and waste mismanagement at End of Life (see Table I.2 for the definition of parameters).

Table I.2 summarises the parameters used in Equations I.1 and I.2, and the default values adopted in the *PLP method*. Alternative, more representative values may be applied in the LCA study, provided that they are adequately documented and justified.

**Table I.2.** Parameters, and respective default values, used in the *PLP method* to estimate macro-plastics loss and release due to product littering and waste mismanagement at End of Life (according to Equations I.1 and I.2) <sup>(1)</sup>.

Parameter	Definition	Default value	Source
$M_{prod,EoL}$	Total mass of product leaving the Use stage and entering End of Life (per functional unit)	LCI data	LCI of the product in scope
$LR_{lit}$	Littering Rate <sup>(2)</sup>	Product-specific	Peano et al. (2020); p. 75
MWI	Mismanaged Waste Index <sup>(3)</sup>	9.25% <sup>(4)</sup>	Peano et al. (2020); <i>PLP Sectorial Guidances Generic Data v1.0</i>
$Rel_{ocean}$	Release rate to ocean	Product-specific	Peano et al. (2020); p. 79
$Rel_{frw}$	Release rate to freshwater		
$Rel_{terenv}$	Release rate to terrestrial environment	Product-specific	Peano et al. (2020); p. 79
RedR	Redistribution rate among compartments	100%	Peano et al. (2020); p. 81

<sup>(1)</sup> Alternative, more representative values may be applied in the LCA study, provided that they are adequately documented and justified.

<sup>(2)</sup> Share of product waste subject to littering events.

<sup>(3)</sup> Share of product waste subject to waste mismanagement practices (e.g. direct discharge of waste to waterways, uncollected waste, poor management, etc.).

<sup>(4)</sup> EU-average value calculated as weighted average of country-specific default values reported in Peano et al. (2020) for single EU Member States. Country-specific values for individual Member States should be applied, where relevant, depending on the geographical scope of the LCA study.

### I.3.3. Micro-plastics loss and release from tire abrasion during transport

Micro-plastics loss and emissions from tire abrasion are calculated as a share of the tire and road wear particles (TRWP) generated due to the friction between tires and pavement during road transport. Specific calculation rules for both tire-related and non-tire-related studies are provided in the *PLP method*. Considering the focus of the present method on plastic products, the calculation approach for non-tire-related studies is summarised in the following.

The mass of tire particles that is considered to be lost as micro-plastics from the technosphere during transport processes (MTA) is calculated based on the total

transported mass ( $m_t$ ; per functional unit), the average vehicle load (AVL), the share of rubber in tire tread (SR) and the loss rate of the tire ( $LR_{tire}$ ), according to Equation I.3. While the transported mass and possibly the average vehicle load are to be derived from the specific product Life Cycle Inventory, the *PLP method* reports default values for the other parameters specifically related to tires (SR and  $LR_{tire}$ ), which depend on the type of vehicle (Peano et al., 2020; pp. 123-124; and Table I.3 below). Alternative values may be applied in the LCA study, provided that they are adequately documented and justified.

$$MTA_i(kg) = MiL_t(kg) = \frac{m_t(kg \cdot km)}{AVL(kg)} * SR(\%) * LR_{tire} \left( \frac{mg}{km} \right)$$

**Equation I.3** – Micro-plastics loss due to tire abrasion during road transport (see Table I.3 for the definition of parameters).

**Table I.3.** Default values applied in the *PLP method* for vehicle-specific parameters required to estimate the loss of micro-plastics from tire abrasion according to Equation I.3 (taken from Peano et al., 2020) <sup>(1)</sup>.

Vehicle type	AVL <sup>(2)</sup> (kg)	SR <sup>(3)</sup> (%)	LR <sub>tire</sub> <sup>(4)</sup> (mg/km)
Heavy truck	12000	60	517
Light truck	3500	36	142
Car	640	35	102
Motorcycles	-	40	45

<sup>(1)</sup> Alternative values may be applied in the LCA study, provided that they are adequately documented and justified.

<sup>(2)</sup> Average Vehicle Load (may be applied in case no supply-chain specific values are available).

<sup>(3)</sup> Share of Rubber in tire treads.

<sup>(4)</sup> Loss Rate of micro-plastics from tires.

Not the entire mass of micro-plastics lost from the technosphere due to tire abrasion ( $MTA_i$ ) is considered to be eventually released to the environment, since the release to the different environmental compartments depends on additional factors including the characteristics of the generated micro-plastics (e.g. size), the location where generation occur (e.g. type of road – rural, urban, highway), and the type of treatment of runoff water containing micro-plastics arising during meteorological events (e.g. retention or transfer to sludge from wastewater treatment). Initial release rates estimated taking into account these factors range from 0% to 66%, depending on the compartment (Peano et al., 2020; p. 133; average values for all types of road and watershed). Possible redistribution of initially released micro-plastics among the different environmental compartments is also taken into account to calculate the actual ultimate release to the ocean ( $MiR_{t,ocean}$ ), freshwater sediments ( $MiR_{t,frw}$ ) and terrestrial environment (including soil;  $MiR_{t,terenv}$ ). This is made by means of redistribution rates, which are used as an approximation to model the partial fate of initially released micro-plastics and calculate final release rates to the mentioned compartments. According to default values used in the *PLP method*, micro-plastics initially released to freshwater are only partially redistributed to the ocean, as 90% is considered to remain in freshwater sediments (i.e. retention rate in freshwater sediments = 90%), while micro-plastics released directly to the ocean and terrestrial environment (including soil) remain in the same compartment (i.e. are not transferred). The final release rates to ocean ( $FinalRelR_{ocean}$ ), freshwater sediments ( $FinalRelR_{frw}$ ) and terrestrial environment ( $FinalRelR_{terenv}$ ) estimated based on

these redistribution rates are reported in Table I.4 (again as average values for all types of road and watershed). These values are used to calculate the ultimate micro-plastics release to such compartments, according to Equation I.4. Alternative, more representative values may be applied in the LCA study, provided that they are adequately documented and justified.

$$MiR_{t,ocean} = MTA_l(kg) * FinalRelR_{ocean} (%) \quad (a)$$

$$MiR_{t,frw} = MTA_l(kg) * FinalRelR_{frw} (%) \quad (b)$$

$$MiR_{t,terenv} = MTA_l(kg) * FinalRelR_{terenv} (%) \quad (c)$$

**Equation I.4** – Micro-plastics release to ocean (a) freshwater sediments (b) and terrestrial environment (including soil) (c) due to tire abrasion during road transport (see Table I.4 for the definition of parameters).

Table I.4 summarises the parameters used in Equations I.3 and I.4, and the default values adopted in the *PLP method*. Alternative, more representative values may be applied in the LCA study, provided that they are adequately documented and justified.

**Table I.4.** Parameters, and respective default values, used in the *PLP method* to estimate micro-plastics loss and release due to tire abrasion (according to Equations I.3 and I.4) <sup>(1)</sup>.

Parameter	Definition	Default value	Source
$m_t$	Total mass of product / material transported (per functional unit)	LCI data	LCI of the product in scope
AVL	Average vehicle load	See Table I.3	Peano et al. (2020); pp. 123-124
SR	Share of rubber in tires		
$LR_{tire}$	Loss rate of micro-plastics from tires		
$FinalRelR_{ocean}$	Final release rate to ocean	2% <sup>(2)</sup>	Peano et al. (2020); pp. 139-140
$FinalRelR_{frw}$	Final release rate to freshwater sediments	15% <sup>(2)</sup>	
$FinalRelR_{terenv}$	Final release rate to terrestrial environment	69% <sup>(2)</sup>	

<sup>(1)</sup> Alternative, more representative values may be applied in the LCA study, provided that they are adequately documented and justified.

<sup>(2)</sup> Average value across different types of road and watershed. Final release rates to ocean, freshwater sediments and terrestrial environment do not add up to 100% as the remaining share (14%) of initially lost tire particles is considered to be taken back in the technosphere as “well managed waste” (i.e. as retentate in wastewater treatment sludge or in storm water management sludge that is eventually landfilled or incinerated).

#### I.3.4. Micro-plastics loss and release from plastic pellets

The *PLP method* provides guidelines to quantify the cumulative loss and release of plastic pellets as micro-plastics along the (upstream) value chain of plastic products (from polymer production/recycling to conversion, including related transport and intermediary services), wherever pellets are used as raw material for product manufacturing (as it is commonly the case for Europe; Hann et al., 2018). However, plastic pellets waste and material loss during the production process are excluded from this quantification, as they are not emitted to the environment (but properly contained and disposed of).

The total mass of plastic pellets cumulatively lost as micro-plastics along the upstream value chain of plastic products ( $MPP_i$ ) is calculated based on the mass of pellets used as input to the product manufacturing process ( $m_p$ ; per functional unit) and the overall loss rate of plastic pellets ( $LR_{pp}$ ), according to Equation I.5. The PLP guidelines recommend to apply a default loss rate in the order of 0.01%, calculated as the average of the range 0.001%-0.1%, derived from the reviewed literature (Peano et al., 2020; p.147). However, considering the high uncertainty affecting this estimate, it is here recommended to apply a value reflecting the high-end of the reported range, i.e. 0.1%, following a conservative approach (Table I.5). This is also in line with the recommended estimate provided in a former study conducted for the Commission (Hann et al., 2018). Alternative values may be applied in the LCA study, provided that they are adequately documented and justified.

$$MPP_i(kg) = MiL_p(kg) = m_p(kg) * LR_{pp}(\%)$$

**Equation I.5** – Micro-plastics loss due to plastic pellets (see Table I.5 for the definition of parameters).

Similarly to micro-plastics generated from tire abrasion, not the entire mass of plastic pellets lost from the technosphere ( $MPP_l$ ) is considered to be eventually released to the environment. Initial release rates accounting for relevant factors affecting actual initial release to a specific environmental compartment are applied, as well as redistribution rates from initial emission compartments to relevant end compartments (i.e. ocean, freshwater sediments and terrestrial environment). These rates are determined considering release and redistribution pathways similar to those assumed for micro-plastics from tire abrasion (Section I.3.3), except for the retention rate in freshwater sediments (set to 30%, based on Hurley et al., 2018)<sup>176</sup>, while the retention rate in the terrestrial environment (including soil) is still assumed equal to 100%. The final release rates (FinalRelR) to ocean, freshwater sediments and terrestrial environment estimated based on such rates and pathways are equal to 12%, 5% and 69%, respectively, with the remaining 14% being recuperated by the wastewater treatment system and eventually routed to landfilling or incineration (Table I.5). These values are applied to calculate the ultimate micro-plastics release to ocean ( $MiR_{p,ocean}$ ), freshwater sediments ( $MiR_{p,frw}$ ) and terrestrial environment ( $MiR_{p,terenv}$ ), following Equation I.6. Alternative, more representative values may be applied in the LCA study, provided that they are adequately documented and justified.

$$MiR_{p,ocean} = MPP_l(kg) * FinalRelR_{ocean} (%) \quad (a)$$

$$MiR_{p,frw} = MPP_l(kg) * FinalRelR_{frw} (%) \quad (b)$$

$$MiR_{p,terenv} = MPP_l(kg) * FinalRelR_{terenv} (%) \quad (c)$$

**Equation I.6** – Micro-plastics release to ocean (a), freshwater sediments (b) and terrestrial environment (including soil) (c) due to plastic pellets (see Table I.5 for the definition of parameters).

Table I.5 summarises the parameters used in Equations I.5 and I.6, and the default values adopted in the *PLP method* or recommended in this method. Alternative, more representative values may be applied in the LCA study, provided that they are adequately documented and justified.

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<sup>176</sup> Hurley et al. (2018) demonstrated that flooding exported approximately 70% of the micro-plastics load stored on river beds. It was thus estimated that only 30% of micro-plastic particles initially released to freshwater are ultimately stored in freshwater sediments.

**Table I.5.** Parameters, and respective default values, used in the *PLP method* to estimate micro-plastics loss and release due to plastic pellets (according to Equations I.5 and I.6) <sup>(1)</sup>.

Parameter	Definition	Default value	Source
$m_p$	Mass of input plastic pellets used in the product manufacturing process (per functional unit)	LCI data	LCI of the product in scope
$LR_{pp}$	Loss rate of plastic pellets	0.1%	Peano et al. (2020); p. 147 Hann et al. (2018)
$FinalRel_{ocean}$	Final release rate to ocean	12% <sup>(2)</sup>	Calculated based on Peano et al. (2020) <sup>(3)</sup>
$FinalRel_{frw}$	Final release rate to freshwater sediments	5% <sup>(2)</sup>	Calculated based on Peano et al. (2020) <sup>(3)</sup>
$FinalRel_{terenv}$	Final release rate to terrestrial environment	69% <sup>(2)</sup>	Calculated based on Peano et al. (2020) <sup>(3)</sup>

<sup>(1)</sup> Alternative, more representative values may be applied in the LCA study, provided that they are adequately documented and justified.

<sup>(2)</sup> Final release rates to ocean, freshwater sediments and terrestrial environment do not add up to 100% as the remaining share (14%) of initially lost plastic pellets is taken back in the technosphere through the wastewater treatment system and is eventually landfilled or incinerated.

<sup>(3)</sup> *PLP\_Sectorial\_Guidances\_Generic\_data*.

### I.3.5. Micro-plastics loss and release from synthetic textile microfibers

The *PLP method* addresses the potential loss and release of micro-plastics from the abrasion of synthetic textiles and textile products during laundering, which lead to the formation of synthetic microfibers. This can occur both during pre-wash and processing steps applied in textile production, and washing activities performed in the Use stage.

The mass of micro-plastics lost from the technosphere as microfibers during washing of synthetic textiles ( $MTX_i$ ) is calculated based on the mass of textile or textile product being washed ( $m_{tx}$ ; per functional unit), its synthetic content (%), the loss rate of microfibers per wash ( $LR_{tx,wash}$ ) and the number of washes per life-cycle ( $n_{wash}$ ; Equation I.7). Default values recommended by the *PLP method* for these parameters (based on the reviewed literature) are reported in Table I.6. Alternative values may be applied in the LCA study, provided that they are adequately documented and justified.

$$MTX_i(kg) = MiL_{tx}(kg) = m_{tx}(kg) * synthetic_{content}(\%) * LR_{tx,wash}\left(\frac{mg}{kg}\right) * n_{wash}$$

**Equation I.7** – Micro-plastics loss due to abrasion of synthetic textiles during washing (see Table I.6 for the definition of parameters).

Although the *PLP method* recognises that synthetic textile microfibers can be released directly to air and ocean (when no wastewater collection and treatment infrastructure is available), it specifically addresses “indirect” release to the different environmental compartments via the wastewater pathway (which is the most relevant situation for average EU conditions, at least regarding washing during the Use stage). Similarly to micro-plastics generated from tire abrasion and plastic pellets, not the entire mass of synthetic textile microfibers lost from the technosphere ( $MTX_i$ ) is considered to be eventually released to the environment. Initial release rates accounting for relevant factors affecting actual initial release to a specific environmental compartment are

applied, as well as redistribution rates from initial emission compartments to relevant end compartments (i.e. ocean, freshwater sediments and terrestrial environment). The PLP guidelines provide a detailed description of the assumed release and redistribution pathways for each environmental compartment, as well as default release and redistribution rates reflecting global average conditions (Peano et al., 2020; pp. 94-95), as reported in Table I.6. Based on these parameters, the ultimate micro-plastics release to ocean ( $MiR_{tx,ocean}$ ), freshwater sediments ( $MiR_{tx,frw}$ ), and terrestrial environment ( $MiR_{tx,terenv}$ ) is estimated according to Equation I.8. Note, however, that wastewater treatment efficiencies (affecting initial release rates) may even largely vary across different (EU) countries, so that country-specific values for individual Member States should be applied, where relevant, instead of those reported in Table I.6, depending on the geographical scope of the LCA study<sup>177</sup>. Alternatively, more representative values (at global or country level) may be applied in the LCA study, provided that they are adequately documented and justified.

$$MiR_{tx,ocean} = MTX_l(kg) * (RelR_{ocean} (%) * RedR_{ocean_{ocean}} (%) + RelR_{frw} (%) * RedR_{frw_{ocean}} (%) + RelR_{soil} (%) * RedR_{soil_{ocean}} (%)) \quad (a)$$

$$MiR_{tx,frw} = MTX_l(kg) * (RelR_{frw} (%) * RedR_{frw_{frw}} (%) + RelR_{soil} (%) * RedR_{soil_{frw}} (%)) \quad (b)$$

$$MiR_{tx,terenv} = MTX_l(kg) * (RelR_{terenv} (%) * RedR_{terenv_{terenv}} (%) + RelR_{soil} (%) * RedR_{soil_{soil}} (%)) \quad (c)$$

**Equation I.8** – Micro-plastics release to ocean (a), freshwater sediments (b), and terrestrial environment (including soil) (c) due to abrasion of synthetic textiles during washing (see Table I.6 for the definition of parameters).

Table I.6 summarises the parameters used in Equations I.7 and I.8, and the default values adopted in the *PLP method*. Alternatively, more representative values may be applied in the LCA study, provided that they are adequately documented and justified.

<sup>177</sup> Default values at single country level adopted in the *PLP method* are available in the accompanying spreadsheet "*PLP\_Sectorial\_Guidances\_Generic\_data*".

**Table I.30.** Parameters, and respective default values, used in the *PLP method* to estimate microplastics loss and release due to abrasion of synthetic textiles during washing under global average conditions (according to Equations I.7 and I.8) <sup>(1)</sup>.

Parameter	Definition	Default value	Source
$m_{tx}$	Mass of textile or textile product being washed (per functional unit)	LCI data	LCI of the product in scope
Synthetic <sub>content</sub>	Synthetic fibre content of the textile (product)	LCI data	LCI of the product in scope
$LR_{tx,wash}$	Loss rate of microfibers per wash ( $LR_{tx,wash}$ )	46 mg/kg <sub>textile</sub> /wash	Peano et al. (2020); pp. 94-95
$n_{wash}$	Number of washes of the textile (product) per life-cycle	20	
$ReIR_{ocean}$	Release rate to ocean	5%	
$ReIR_{frw}$	Release rate to freshwater	14%	
$ReIR_{terenv}$	Release rate to terrestrial environment	0%	
$ReIR_{soil}$	Release rate to soil	41%	
$RedR_{ocean\_ocean}$	Redistribution rate within ocean	100%	
$RedR_{frw\_ocean}$	Redistribution rate from freshwater to ocean	70%	
$RedR_{soil\_ocean}$	Redistribution rate from soil to ocean	51%	
$RedR_{frw\_frw}$	Redistribution rate within freshwater	30%	
$RedR_{soil\_frw}$	Redistribution rate from soil to freshwater	22%	
$RedR_{terenv\_terenv}$	Redistribution within terrestrial environment	100%	
$RedR_{soil\_soil}$	Redistribution rate from soil to terrestrial environment	27%	

<sup>(1)</sup> Country-specific values for individual Member States should be applied, where relevant, instead of global average values reported in this table, depending on the geographical scope of the LCA study. Default values at single country level adopted in the *PLP method* are available in the accompanying spreadsheet "*PLP\_Sectorial\_Guidances\_Generic\_data*". Alternative, more representative values (at global or country level) may be applied in the LCA study, provided that they are adequately documented and justified.

### I.3.6. Summary of plastic loss and release indicators

Table I.7 provides a summary of the indicators that may be calculated through the *PLP method* in relation to macro- and micro-plastics generation (loss from the technosphere) and release to the environment in different end compartments (ocean and terrestrial environment).

**Table I.7.** Indicators of macro- and micro-plastics loss (generation) and release to the environment that may be calculated through the *PLP method*.

Type of plastic	Indicator	Calculation
Macro-plastics	Loss (at End of Life)	$MaL = (MaL_l + MaL_{wm}) =_l MPW^{(1)}$
	Release to ocean	$MaR_{ocean}^{(2)}$
	Release to terrestrial environment	$MaR_{terenv}^{(2)}$
Micro-plastics	Loss <sup>(3)</sup>	$MiL = MiL_t + MiL_p + MiL_{tx} = MTA_l + MPP_l + MTX_l$
	Release to ocean <sup>(4)</sup>	$MiR_{ocean} = MiR_{t,ocean} + MiR_{p,ocean} + MiR_{tx,ocean}$
	Release to freshwater sediments <sup>(4)</sup>	$MiR_{frw} = MiR_{t,frw} + MiR_{p,frw} + MiR_{tx,frw}$
	Release to terrestrial environment (including soil) <sup>(4)</sup>	$MiR_{terenv} = MiR_{t,terenv} + MiR_{p,terenv} + MiR_{tx,terenv}$

<sup>(1)</sup> Inherently accounts for the contribution of macro-plastics loss at End of Life due to both product littering ( $MaL_l$ ) and waste mismanagement ( $MaL_{wm}$ ).

<sup>(2)</sup> Calculated based on the total macro-plastics loss at End of Life ( $MaL$ ), through specific release and redistribution rates among compartments. Inherently accounts for the combined contribution of product littering and waste mismanagement.

<sup>(3)</sup> Total micro-plastics loss throughout the product life cycle, accounting for the contribution of sources occurring at different life cycle stages, i.e. tire abrasion during transport ( $MiL_t$ ;  $MTA_l$ ), pellet losses from relevant upstream processes ( $MiL_p$ ;  $MPP_l$ ), and abrasion of synthetic textiles during washing (e.g. during Use;  $MiL_{tx}$ ;  $MTX_l$ ).

<sup>(4)</sup> Total micro-plastics release accounting for the contribution of the same sources considered for the calculation of the total micro-plastics loss.

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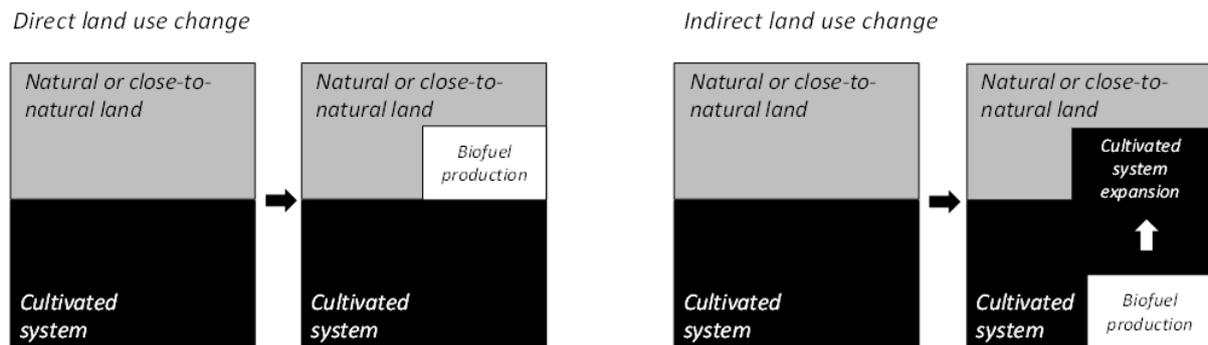
## **Annex J - Land use changes: background information and models available to quantify GHG emissions from iLUC in LCA**

This Annex provides additional background information and discussion on land use changes and on the modelling of indirect land use change in a LCA context. In particular, section J.1 provides broader definitions of both direct and indirect Land Use Change (dLUC/iLUC) compared to those given in the main body of this document (Section 4.4.15.3), while Section J.2 reports an overview of models currently available to quantify GHG emissions from iLUC in LCA. Finally, section J.3 describes more in detail the iLUC model presented by Schmidt et al. (2015), and the main calculation steps that should be conducted when applying it to quantify GHG emissions from iLUC for a product using biomass as feedstock. The model may be applied to provide alternative estimates of the iLUC contribution of the product in scope to the Climate Change impact indicator, when calculating additional environmental information. However, any alternative estimate shall not replace the iLUC contribution calculated based on recommended default emission factors reported in Section 4.4.15.3 (Table 16), and shall be reported separately in addition to the former contribution.

### **J.1. Land use changes: direct and indirect (dLUC/iLUC)**

Different but somehow aligned definitions of dLUC and iLUC exist in the literature. ISO 14067:2018 defines direct land use change (dLUC) as a "change in human use or management of land within the boundaries of the product system being assessed", while iLUC as "change in the use or management of land which is a consequence of direct land use change, but which occurs outside the product system being assessed". Another study by Schmidt et al. (2015) defines dLUC as "those changes that occur on the same land as the land use", and iLUC as "the upstream life cycle consequences of the land use regardless of the purpose of the land use", i.e. a change in land use caused indirectly as an upstream consequence of a dLUC but taking place somewhere else in the World. Marelli et al. (2015) define a land use change to be direct when "the demanded crops are grown on uncultivated land", while indirect "when the demanded crops are grown on already cultivated or used land". In the scientific literature, dLUC has also been defined as "all changes in above- and below-ground flows of carbon, and nitrogen and phosphorus flows on a particular site, as one land use takes place instead of another" (e.g. Hamelin et al., 2012; Tonini et al., 2012). In a nutshell, it can be stated that dLUC refers to the changes occurring on the same land where the land use for the product under assessment takes place. The iLUC refers instead to market-driven consequences incurred (somewhere else) by the dLUC taking place in the very first place (Figure J.1). The point of departure for iLUC to occur is when arable land, already-in-use for cropping or grazing activities, is used for supplying the feedstock under assessment. In other words, iLUC arises as changes in overall land demand occur. The pre-condition for iLUC to occur is that the global agricultural area is still expanding because of increased population, GDP (Gross Domestic Product) increase of some countries, and that its availability is inherently limited/constrained. For example, if the feedstock needed for a bio-based product or biofuel is cultivated at the expense of another crop, the service this formerly-cultivated crop provided to the food/feed market will still be demanded on the World's market. The main underlying postulate of iLUC is that this relative drop in supply is likely to cause a relative increase in agricultural prices, which in turn provides incentives to increase production elsewhere. This in principle can incur: i) agricultural land expansion (at the expenses of nature), ii) intensification of (agricultural) production, and iii) crop-displacement mechanisms (reduced consumption). The latter is supported by some studies arguing that in the short-to-medium term not 100% of the displaced feedstock may need to be compensated by increased production as reduced consumption may also occur (e.g. Edwards et al., 2010). This hypothesis is however contrasted by other authors (e.g. Schmidt et al., 2015), arguing that this effect should not be included in LCAs, since it is the long-term effect of the demand that should be guiding for decisions (Weidema et al., 2013). According to these authors, the supply of goods and

services should be assumed fully elastic, i.e. an increase in demand is to be met by a corresponding (1:1) increase in supply.



**Figure J.1.** Schematic representation of direct and indirect land use changes considering biofuel production as an example. Adapted from CE Delft (2010).

## J.2. Overview of models available for quantification of iLUC GHG emissions

A number of approaches and models have been proposed in recent years to quantify iLUCs in LCA, but a broad consensus on what to apply still needs to be reached (Warner et al., 2014). According to De Rosa et al. (2016), the main challenges are: i) the identification of the affected (in consequential LCA often referred as to marginal) land; ii) establishing the relationship between the demand for agricultural products and the occurring land use changes; iii) including the effect of by-products; iv) the overall level of uncertainty caused by the multiple modelling assumptions involved as highlighted in Marelli et al. (2011). The LUC models are typically distinguished into 3 types (De Rosa et al., 2016): *economic equilibrium models* (EEMs), *causal-descriptive models* (CDMs) and *role-based normative models* (NMs).

EEMs are based on the theory of economic equilibrium: changes in supply and demand induce fluctuations of the price of goods until a new equilibrium “supply=demand” is reached, with a new price. Any variation in goods demand incurs changes in land requirement and occupation from which LUC between the former and new equilibriums can be estimated. Two main types of EEMs models exist: partial equilibrium models (PE), restricting the modelling to selected sectors of the economy (e.g. CAPRI, 2012), and computable general equilibrium models (CGE) striving to include and link all the sectors of the global (or regional) economy (e.g. among others, GTAP, IMAGE, LEITAP; see Britz and Hertel, 2011).

CDMs (sometimes also referred to as biophysical or deterministic) are simpler and conceptually easier than EEMs (Nassar et al., 2011), reducing computational efforts and data needs. They describe future states of a system establishing cause-effect relationships. These can be determined from a combination of biological and physical land characteristics, own and cross-price elasticities, statistical data, etc. (De Rosa et al., 2016). CDMs do not exclude economic aspects that drive the supply-demand patterns; rather, they forecast future production and consumption patterns based on current or historical market trends and assumptions on agriculture supply-demand trajectories. Based on this, future land uses and their geographic origin (i.e. land areas affected by a change in demand/supply) can be identified. Recent examples of CDMs are those described in Bauen et al. (2010), Schmidt et al. (2015), and Tonini et al. (2016).

Normative models attempt to establish assumptions or LUC factors based on statistical metadata (Audsley et al., 2009). Often, they de facto exclude iLUC, avoiding the most controversial aspect, and only focus on the quantification of dLUCs GHG emissions; an example is the approach proposed in the PAS 2050 (BSI, 2011), and that from Flynn et al. (2012). Another NM is that proposed by Fritsche et al. (2010), where the bioproduct is assumed to come by 25% from set-free land with no iLUC risk, and by 75% from new

land incurring iLUCs. The iLUC GHG factors reported in EU 2015/1513 (EC, 2015) can also be classified under this category. In EU 2015/1513, a number of default iLUC GHG factors are provided for biofuels obtained from sugar-, starch-, and oil-rich crops. The figures were derived from a meta-analysis of the iLUC GHG factors reported in the scientific literature and are originally reported as g CO<sub>2</sub>-eq. MJ<sup>-1</sup>, as intended to be applied to biofuels assessment studies. An overview of the main differences between EEMs (e.g. Valin et al., 2015) and CDMs (e.g. Schmidt et al., 2015) is reported in Table J.1.

**Table J.1.** Main differences between EEMs (economic equilibrium models) and CDMs (causal descriptive models).

	<b>EEMs</b> <i>(e.g. Valin et al., 2015)</i>	<b>CDMs</b> <i>(e.g. Schmidt et al., 2015; Tonini et al., 2016; Bauen et al., 2010)</i>
<b>Model type</b>	EEM (Global economic equilibrium model) targeted to economic analyses	CDM (causal-effect/deterministic) specifically targeted to application in LCA
<b>Type of iLUC factor</b>	Crop- and location-specific	Global, independent on crop type and location of land occupation
<b>What iLUC includes?</b>	iLUC + all the substitution of co-products at farm and biorefinery level (in terms of land avoided)	iLUC only
<b>Land suppliers</b>	Transformation, intensification, reduced consumption	Transformation, intensification
<b>Identification of affected lands (i.e. marginal)</b>	Estimated using price and price elasticities that are implemented as model functions	Estimated with historical patterns using transformation matrices that can be changed/updated (mainly based on FAO data)
<b>Intensification effects</b>	Accounted for as reduced land needs, but without accounting for associated GHG impacts due to increased fertilisers use	Accounted for both as reduced land needs and as GHG emissions due to increased fertilisers use
<b>Assessment of GHG emissions from LUC-deforestation</b>	Annual amortisation of the initial C-emissions using a 20yr period	<u>Schmidt et al. (2015)</u> : After 1-year of occupation, land is released back to other product systems (uses). This equals to speed up the emission by one year. Follows using Bern C-Cycle and IPCC-GWP (Forster et al., 2007) to calculate the change in radiative forcing <u>Tonini et al. (2016)/Bauen et al. (2010)</u> : Annual amortisation
<b>Previous applications</b>	A number of studies on biofuels (e.g. Valin et al., 2015)	A number of Danish Energy Agency, Danish EPA, private-companies and peer-review studies on food, biofuels, and bioproducts

Pros and cons of these models, in the endeavour of their application to LCA, have been highlighted in two recent reviews by Warner et al. (2014) and De Rosa et al. (2016). In particular, De Rosa et al. (2016) performed a pairwise comparison, based on a number of criteria, between: a PE model (CAPRI, 2012), a CGE model (GTAP-AEZ), a hybrid CGE/PE model developed by JRC (integrating data from the CGE model IFPRI-MIRAGE and the PE model AGLINK-COSIMO; Hiederer et al., 2010), a NM (BSI, 2011), and two CDMs (Bauen et al., 2010; Schmidt et al., 2015). The criteria of the comparison were: i) *completeness of scope*, ii) *impact assessment relevance*, iii) *scientific robustness and certainty*, and iv)

*transparency, reproducibility and applicability.* The main results of the analysis are summarised below.

**Completeness of scope:** generally, GTAP-AEZ and JRC have more complete datasets and land classification maps to derive the origin of the affected (marginal) land compared to CDMs. Yet, the uncertainty around this is high as it depends upon the assumptions regarding the competition for land; this is described by the function of elasticity of land transformation  $\sigma$  (Hertel et al., 2009). Land transformation elasticity distributes the productivity-adjusted land to its alternative uses. Regarding the distribution of the GHG emissions, only Schmidt et al. (2015) model suggests a methodology for handling this. Other EEMs leave this aspect to the users or consider the issue outside the scope of LUC modelling. EEMs have usually a (rough) national level of GHG data aggregation. A common limitation of all LUC models, regardless of the approach and resolution of GHG emissions, is the amortisation of the emission over an arbitrary period of time, generally 20 years (Fritsche et al., 2010; Valin et al., 2015) or 30 years (Bauen et al., 2010). Schmidt et al. (2015) propose an alternative approach that avoids *in toto* amortisation.

**Impact assessment relevance:** all reported models only focus on GHG emissions and lack a broader coverage of the iLUC environmental effects.

**Scientific robustness and certainty:** all reported models do not assess nor propagate uncertainties. Regarding updates, GTAP-AEZ and CAPRI are constantly updated. Schmidt et al. (2015) is updated with a biannual frequency. Bauen et al. (2010) and JRC provide suggestions for future developments, but are not regularly updated. PAS 2050 has not been updated. PAS 2050 and Bauen et al. (2010) are the only ones that have not been peer-reviewed.

**Transparency, reproducibility, and applicability:** All reported models are well documented. CAPRI and JRC have a limited focus on agriculture, their analyses mostly focus on biofuels, and a regional scope (EU) only is available. Bauen et al. (2010) is limited to biofuels analyses. In contrast, GTAP-AEZ, Schmidt et al. (2015) and PAS 2050 models are designed to be applied regardless of location and economic sector, thus having a larger applicability. It should be borne in mind that in LCA it is common practice to assume long-term full elasticity of supply under a competitive unconstrained market. In this respect, EEMs-derived results may instead reflect fluctuations of market prices due to a short-term inelastic supply where a sudden demand increase (or supply decrease) induces a higher price (new equilibrium) and vice versa. This may ultimately generate incongruences in the LCA.

The main conclusions from the reviews of Warner et al. (2014) and De Rosa et al. (2016), can be summarised as follows:

- CDMs specifically developed for application to LCAs may be more suitable than applying more complex and computation-wise intensive EEMs.
- EEMs, however, appear useful for identifying the affected (i.e. marginal) land.
- EEMs address iLUC by attempting to capture all crop displacements at the specific crop and country level (including intensification). EEMs identify the affected crops by price and price elasticity information, and specific crop markets are assumed (e.g. rapeseed displaced in one country is compensated with the quantity of another crop producing an equivalent amount of oil, using elasticities of substitutions). This leads to crop- and country-specific iLUC factors.
- EEMs describe land competition and transformation through mathematical functions (elasticities), difficult to calibrate with real data. When calibration is made, it is not free from uncertainties and may ultimately only reflect case-specific scenarios.
- EEMs may include the effect of co-products, potentially incurring double counting in LCAs; these should not be included in LUC models for LCAs, as they belong to a part of the product system not related to the actual provision of land.

- The iLUC factors derived with EEMs represent the sum of direct and indirect effects; in principle, double counting (e.g. for dLUC) should be avoided.
- A careful combination of the two modelling approaches (CDMs and EEMs), whenever possible, is ultimately encouraged.
- Time allocation of GHG emissions over an arbitrary period of time (e.g. 20yr) should be avoided.
- To avoid arbitrary allocation, alternative time-dynamic formulations exist (among the others see: (Cherubini et al., 2011, 2016; Kloverpris and Mueller, 2013; Schmidt et al., 2015).
- Incongruences may be generated when using EEMs, as their results reflect prices fluctuations following short-term inelasticity of supply, while LCA typically assumes long-term full elasticity of supply.

### **J.3. Alternative model applicable to quantify iLUC GHG emissions to calculate additional environmental information**

This section describes the main features of the model proposed by Schmidt et al. (2015) as a conceptual framework for a consistent modelling of iLUC in LCA. The main calculation steps required to apply the model to quantify the iLUC GHG emissions associated with the use of a specific biomass in a product are also reported.

#### **J.3.1. General information on the model**

The model proposed by Schmidt et al. (2015) belongs to the category of CDMs and differs from EEMs for a number of aspects (see Section J.2), mainly:

1. No amortisation of the GHG emissions from deforestation is performed as a time-dynamic approach is instead applied.
2. A full elasticity of the supply is considered, i.e. the demand for one product unit will in the long-term lead to supply of one more product unit, without incurring reduction in the consumption (Weidema, 2003). This is valid under a competitive unconstrained market where long-term market prices are determined by the long-term marginal production costs. In the iLUC model, this assumption implies that a marginal change in demand for land does not have any long-term effects on prices of land-based products. According to Schmidt et al. (2015), this is confirmed by the food commodity prices that over the period 1961-2014 have not increased relative to the general consumer index, while recent short-time price peaks for food products can largely be explained by changes in fuel prices and speculation.
3. The market for land is considered to be global; five specific land markets are considered.

#### **J.3.2. Markets for land considered in the model**

In Schmidt et al. (2015), the land is considered as “capacity for biomass production”, i.e. a capital input. Regional differences in land productivity are handled using the net primary production (NPP<sub>o</sub>) figures from Haberl *et al.*, (2007). The relative productivity of a specific region is calculated as its NPP<sub>o</sub> divided by the world average NPP<sub>o</sub> (6110 kg C ha<sup>-1</sup>y<sup>-1</sup>; Haberl *et al.*, 2007). This means that, while the iLUC factor expressed per hectare for lands with different productivity will be the same (see earlier), the iLUC factor expressed per unit of crop grown on these lands will differ according with their relative productivity.

The model from Schmidt et al. (2015) considers five different global markets for land, representing all land types in the world: i) arable land (fit for annual and perennial crops, intensive/extensive forestry, and pasture); ii) intensive forest land (fit for intensive/extensive forestry and pasture, but not for cropping), iii) extensive forest land

(not fit for intensive because too hilly, remote, uneconomic, unfertile, etc.), iv) grassland (fit for pasture, not for cropping and forest, being too dry), v) barren land (not fit for biomass production). Each market can be supplied with different inputs of land already in use, intensification, and transformation. The inputs to the individual markets (e.g. to 1 ha of arable land) are quantified based on historical patterns. This also applies for the ratio between the share of land provided through intensification of current production and that provided through expansion on forestland (deforestation). The reader is referred to Figure J.2 and Schmidt et al. (2015) for more details.

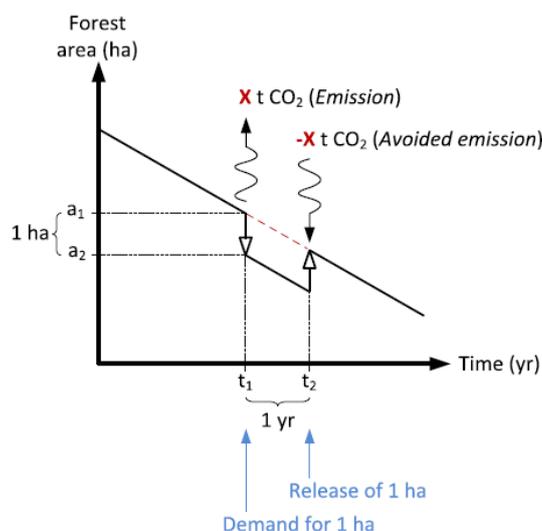
**Table 1**  
Included markets for land. The markets represent the marginal potential uses of the land and are independent of the actual land cover.

Markets for land	Inputs to the markets	Description
Market for arable land (fit for arable and other)	<ul style="list-style-type: none"> <li>• Cropland already in use</li> <li>• Transformation of land not in use to cropland</li> <li>• Intensification of cropland already in use</li> </ul>	Fit for arable cropping (both annual and perennial crops), for intensive or extensive forestry, and pasture.
Market for intensive forest land (fit for intensive/extensive forestry and grazing)	<ul style="list-style-type: none"> <li>• Intensive forest land already in use</li> <li>• Transformation of primary/secondary forest (&gt;15% tree cover) to extensive forest</li> </ul>	Fit for intensive forestry but unfit for arable cropping e.g. because the soil is too rocky. Forest crops grown on intensive forest land may be managed intensively or extensively. Intensive forest land may also be used for other uses, e.g. livestock grazing and extensive forestry.
Market for extensive forest land (fit for extensive forestry and grazing)	<ul style="list-style-type: none"> <li>• Extensive forest land already in use</li> <li>• Transformation of primary forest (&gt;15% tree cover) to extensive forest</li> </ul>	Not fit for more intensive forestry (with clear cutting and reforestation, species control, etc.) e.g. because it is too hilly, too remote, or very infertile, making intensive forestry uneconomic. Forests grown on extensive forest lands are typically harvested after natural regrowth with mixed species. Too dry for forestry and arable cropping. Grassland is most often used for grazing.
Market for grassland (fit for grazing)	<ul style="list-style-type: none"> <li>• Grassland already in use</li> <li>• Transformation of grassland not in use to grassland in use</li> </ul>	
Market for barren land (not fit for biomass production)	<ul style="list-style-type: none"> <li>• Barren land already in use</li> <li>• Transformation of barren land not in use to barren land in use</li> </ul>	Too dry for any biomass production.

**Figure J.2.** Markets for land considered in the iLUC model proposed by Schmidt et al. (2015). Taken from Schmidt et al. (2015).

### J.3.3. Handling of carbon emissions from deforestation (dynamic accounting)

Schmidt et al. (2015) models the actual acceleration of deforestation and related CO<sub>2</sub> emissions, and therefore does not need to apply an arbitrary amortisation period (e.g. 20 years). If only expansion is considered, occupation of 1 ha in 1 year will cause 1 ha deforestation. After the 1-year occupation, the land is released back to the market for land, i.e. to other crops, which can then be grown without deforestation (Figure J.3). From a modelling perspective, this equals considering deforestation at time  $t_1$  and an avoided deforestation at  $t_2$ , i.e. speeding up clearing of that hectare by one year. This time-shifted deforestation is valid under the assumption of a general net deforestation trend (and net expansion of managed lands). If this trend stops, the modelling should be changed as a delayed relaxation of natural areas. Note that this approach was also proposed in Kloverpris and Mueller (2013). Knowing the amount of stock cleared (above plus below ground and possible soil organic carbon loss), the Global Warming Potential of bringing land clearing one year forward can be calculated applying the Bern Carbon Cycle and the formula for GWP from Forster et al. (2007). Accordingly, the GWP (over a 100y period) of clearing 1 ha of land at year  $i-1$  (instead of  $i$ ) equals 0.992 kg CO<sub>2</sub>-eq. kg<sup>-1</sup> CO<sub>2</sub>; in other words, the increased global warming effect of speeding up by one year equals  $1-0.992=0.0076$  kg CO<sub>2</sub>-eq.



**Figure J.3.** Effect of LUC on the timing of CO<sub>2</sub> emissions from deforestation, as applied in the iLUC model proposed by Schmidt et al. (2015). Taken from Schmidt et al. (2015).

#### J.3.4. Steps needed to quantify the iLUC GHG contribution

To quantify the iLUC GHG emission associated to the demand for a certain quantity of land (or, alternatively, the demand for a certain amount of biomass) for a product, the following steps are to be followed:

- **Step 1:** Identify the **quantity of land occupation** (in ha\*year/t<sub>biomass</sub>, based on crop yield).
- **Step 2:** Identify the **location of the occupation** (region, e.g. EU27).
- **Step 3:** Derive the **potential net primary production (NPP<sub>0</sub>)** at this location (in t<sub>biomass</sub> C/ha/y; e.g. using maps provided in Haberl et al., 2007 or the dataset in <https://lca-net.com/clubs/iluc>).
- **Step 4:** Specify the **potential use of the occupied land**: (arable, intensive forest, extensive forest, grassland, barren land).
- **Step 5:** Quantify the **productivity factor** (Productivity Factor = NPP<sub>0</sub>/world average NPP<sub>0</sub>); world average NPP<sub>0</sub> (arable land) = 5.68 t C/ha\*year.
- **Step 6:** Convert the actual occupied area (ha\*year/t<sub>biomass</sub>) to **units of productivity weighted hectare years** (pw ha\*year/t<sub>biomass</sub>): pw ha\*year/t<sub>biomass</sub> = Land occupied (ha\*year/t<sub>biomass</sub>) \* Productivity Factor (pw ha\*year/(ha\*year)).
- **Step 7:** Calculate the **iLUC GHG contribution**: iLUC GHG = productivity weighted hectare years (pw ha\*year/t<sub>biomass</sub>) \* iLUC factor (kg CO<sub>2</sub>/pw ha\*year); the iLUC factor should reflect the potential use of the occupied land (Step 4).

The iLUC factors (for arable land, intensive and extensive forest, grassland, barren) are obtained as a combination of deforestation and intensification and are reported in the original iLUC model by Schmidt et al. (2015).

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## **Annex K - Systematic review of selected LCA studies on plastics**

As a preliminary step, a systematic review of selected existing studies developed at the European and international level related to LCA in the field of plastics was performed, to inform and collect potentially useful inputs to develop the *Plastics LCA* method and the related supporting case studies. The specific objectives of the review were to:

- Identify key methodological/modelling aspects and challenges when conducting LCAs of plastic products;
- Review the methodological choices and approaches commonly applied to address such aspects, focusing on the most challenging and controversial ones;
- Evaluate whether uniform and harmonised choices/approaches were used, or if (further) harmonisation and improved consistency were needed;
- Map the base of existing case studies to understand the availability of life cycle inventory data on the covered products, polymers and key intermediates;
- Highlight possible knowledge/methodological gaps and research needs.

### **K.1. Approach**

Potentially relevant studies to be possibly considered in the systematic review were initially collected through multiple targeted literature searches, as described in Section K.1.1. Collected studies were then classified and screened, by recording information related to a number of general aspects (Section K.1.2). These were useful to identify a more limited number of relevant studies to be reviewed in-depth, based on a range of selection criteria (Section K.1.3), and to map initial data availability for developing the supporting illustrative case studies. An in-depth assessment of selected studies was finally performed, focusing on different methodological and modelling aspects, as detailed in Section K.1.4.

#### **K.1.1. Collection of potentially relevant studies**

Potentially relevant peer-reviewed scientific studies focusing at least on one alternative feedstock or material for plastic products and published during the timeframe 2008-2018 were initially searched via extended *Scopus* searches, using the following keywords:

- (bioplastic OR bio-plastic) AND (LCA OR life cycle assessment);
- (biobased plastic OR bio-based plastic) AND (LCA OR life cycle assessment);
- biodegradable plastic AND (LCA OR life cycle assessment);
- (biopolymer OR bio-polymer) AND (LCA OR life cycle assessment);
- (biobased OR bio-based polymer) AND (LCA OR life cycle assessment);
- (recycled or recovered) AND plastics AND (LCA OR life cycle assessment);
- (CO<sub>2</sub> or CO<sub>2</sub>-based) AND (plastic or polymer) AND (LCA OR life cycle assessment).

Potentially relevant technical reports were then searched through traditional web search engines using the same keywords. Cross-check searches were also performed via *Scopus*, considering the names of relevant polymers and monomers from alternative feedstocks and related key intermediates currently available on the market, within the search key: name of polymer/monomer/key intermediate AND (LCA or life cycle assessment). The following bio-polymers were specifically considered (with both full name and respective acronym): bio-PE (Polyethylene), bio-PET (Polyethylene Terephthalate), bio-PP (Polypropylene), bio-PVC (Polyvinylchloride), bio-PUR (Polyurethane), PEF (Polyethylene Furanoate), starch blends, TPS (Thermoplastic Starch), PLA (Polylactic Acid), PHAs (PolyHydroxyAlkanoates), PBS (PolyButylene Succinate), PBSA (PolyButylene Succinate Adipate), PBAT (PolyButylene Adipate co-Terephthalate), PTT (PolyTrimethylene Terephthalate). For bio-monomers and key

intermediates, the search included bio-ethylene, bio-ethylene glycol, bio-butylene, furandicarboxylic acid, lactic acid, hydroxyalcanoic acids; succinic acid, 1,4-butanediol, adipic acid, and 1,3-propanediol. Recycled polymers were also considered in cross-check searches, including recycled PET (Polyethylene Terephthalate), PE (Polyethylene), PP (Polypropylene), PS (Polystyrene), EPS (Expanded Polystyrene), PVC (Polyvinylchloride), and considering again their full name ("recycled + polymer name") or acronym ("R-polymer name") as a search criteria. Finally, CO<sub>2</sub>-based polymers were considered, following the same logic, and especially focusing on CO<sub>2</sub>-based olefins (PE and PP) and CO<sub>2</sub>-based Polyurethane (PUR).

Overall, 171 documents were collected through this procedure (see the full list at the end of this Annex), subdivided into the following main categories of product/study:

- Monomers/Intermediates (34);
- Polymers (58);
- Plastic products/articles (65);
- End-of-life related studies (14).

CEN standards have been also analysed in detail, and taken into account as appropriate in the development of the draft method. However, due to their different (prescriptive) nature, or their primary focus on other aspects than specific methodological rules, they have not been specifically considered for the purpose of this review, and thus are not included in the 171 documents categorized above. In particular, we focused on the following EN standards:

- EN 16760(2015) Bio-based products – Life Cycle Assessment;
- CEN TR 16957(2016) Bio-based products – Guidelines for Life Cycle Inventory (LCI) for the End-of-life phase;
- EN 13432(2000) Packaging - Requirements for packaging recoverable through composting and biodegradation - Test scheme and evaluation criteria for the final acceptance of packaging;
- EN 14995(2006) Plastics - Evaluation of compostability - Test scheme and specifications;
- EN 17033(2018) Plastics - Biodegradable mulch films for use in agriculture and horticulture - Requirements and test methods.

### **K.1.2. Screening assessment**

A screening assessment of all 171 retrieved documents was performed, to collect basic information useful for classification and selection of relevant studies for a more in-depth assessment of methodological choices and data availability (Section K.1.3). The following information was specifically collected in the screening assessment:

- Goal of the study;
- Analysed product(s) and related polymer(s);
- Comparative or non-comparative study (Y/N);
- Type of feedstocks used (including the type of biomass used, if relevant and available);
- Geographical scope;
- Analysed product/end-of-life scenarios (if any);
- Type of modelling approach (attributinal, consequential or not specified);

- System boundary (i.e. “cradle-to-grave”, “cradle-to-gate + grave”, “cradle-to-gate”, “gate-to-gate” or “End of Life”)<sup>178</sup>;
- Functional unit;
- Number of midpoint impact categories assessed;
- Availability of characterised midpoint impact assessment results (Y/N);
- Availability of supplementary material/information or background report with detailed description of modelling approach and LCI data (Y/N);
- Compliance of the study with ISO 14040/14044 standards (Y/N);
- Critical review of the study according to ISO 14040/14044 standards (Y/N).

### **K.1.3. Selection of relevant studies for in-depth assessment**

To limit the scope of the in-depth assessment to studies with a minimum level of quality and detail, a range of selection criteria was applied:

- At least 2 alternatives are assessed;
- At least one midpoint environmental impact category is evaluated (giving priority to studies covering a broad range of categories; studies addressing only energy demand were excluded);
- Characterised midpoint LCIA results (expressed in physical units) are provided;
- A background report or supplementary material/information adequately detailing the modelling approach and LCI data used is available<sup>179</sup>;
- Overall relevance for the purposes of the project (in terms of goals and scope).

After applying the listed criteria, 32 studies were selected for the in-depth assessment (Table K.1), subdivided according to the following product or study categories:

- Monomers/Intermediates (4);
- Polymers (13);
- Plastic products/articles (13);
- End-of-life related studies (2).

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<sup>178</sup> Note that according to ISO 14040/14044, the system boundary of a LCA study shall cover the whole product life cycle (i.e. only “cradle-to-grave” studies” are conceived). Studies focusing only on some life cycle stages are specific applications falling outside the scope of ISO standards.

<sup>179</sup> Or at least extensive description of methods and data is provided in the study.

**Table K.1.** List of screened studies selected for in-depth assessment in the systematic review.

<b>N.</b>	<b>Reference</b>	<b>Product/study category (1)</b>	<b>Scope / Description</b>
1	Daful et al. (2016)	M/I	Lactic Acid (LA) production from lignocellulosic feedstock (sugarcane bagasse and leaves) in an existing South African sugar mill, and comparison with fossil-based LA
2	Forte et al. (2016)	M/I	Bio-based BDO (1,4-butanediol) production via direct fermentation of renewable sugars from lignocellulosic feedstock (wheat straw) in Italy and comparison with fossil-based BDO
3	Liptow et al. (2013)	M/I	Bio-ethylene production from sawmill chips at industrial scale in Sweden, with focus on the identification of possible environmental key contributors
4	Parajuli et al. (2017)	M/I	Comparison of two standalone biorefinery systems with an integrated biorefinery using winter wheat straw and alfalfa as feedstock for bio-based ethanol and bio-based lactic acid in Denmark (following an attributional and consequential approach)
5	Alvarenga et al. (2013a)	P	Bio-based PVC (Polyvinylchloride) resin production from sugarcane-derived bioethanol in Brazil and comparison with fossil-based PVC (attributional approach)
6	Alvarenga et al. (2013b)	P	Bio-based PVC resin production from sugarcane-derived bioethanol in Brazil and comparison with fossil-based PVC (consequential approach)
7	Belboom and Léonard (2016)	P	Bio-based HDPE (High-Density Polyethylene) production from sugar beet or wheat cultivated in Belgium and comparison with conventional fossil-based HDPE (including also End of Life)
8	Groot and Borén (2010)	P	Production of L-lactide, D-lactide, PLLA (Poly-L-Lactic Acid), and two PLLA/PDLA (Poly-D-Lactic Acid) blends from cane sugar in Thailand and comparison with common fossil-based polymers
9	Hansen et al. (2015)	P	GP-PS and HI-PS (General Purpose and High Impact Polystyrene) production from sugarcane ethanol in Brazil and comparison with their fossil-based counterpart (including also End of Life)
10	Hottle et al. (2017)	P	Drop-in bio-polymers from sugarcane (bio-PET, bio-HDPE, bio-LDPE) and compostable biopolymers from corn (PLA, TPS) production in the US and comparison with traditional fossil-based polymers (PET, HDPE, LDPE), including also End of Life

<b>N.</b>	<b>Reference</b>	<b>Product/study category (1)</b>	<b>Scope / Description</b>
11	Kendall (2012)	P	PHB (PolyHydroxyButyrate) production from cellulose in the organic fraction of material recovery facility (MRF) residuals in the US and comparison with PHB from a dedicated crop -corn- (including also End of Life)
12	Kim and Dale (2008)	P	Assessment of cradle-to-gate non-renewable energy use and greenhouse gas emissions of corn-starch-based PHB production in an real PHB facility in Iowa (US), and comparison with common petroleum-derived polymers
13	Liptow and Tillman (2012)	P	Comparison of sugarcane-based LDPE (Low-Density Polyethylene) and fossil-based LDPE under Swedish and European average conditions (including also End of Life)
14	Posen et al. (2016)	P	Comparison of PLA (Polylactic Acid), PHB and bioethylene-based plastics produced from corn or switchgrass in the US with the 8 highest-volume thermoplastic polymers
15	Shen et al. (2012)	P	Comparison of petrochemical PET (Polyethylene Terephthalate), bio-based PET (from sugarcane and maize), recycled PET, recycled bio-based PET, PLA (from maize and sugarcane) and man-made cellulosic fibre from wood pulp, as polymers in primary form, bottles and/or fibres, under (western) European conditions (considering also End of Life)
16	Tsiropoulos et al. (2015)	P	Bio-based HDPE and partially bio-based PET production from Brazilian and Indian sugarcane ethanol and comparison with their petrochemical counterparts under European conditions
17	Van Uytvanck et al. (2014)	P	Bio-based PET production from sugarcane- or willow-derived ethylene in the UK compared to fossil-based PET
18	Arnold and Alston (2012)	P/A	Manufacture, use and End of Life of PP (Polypropylene) tree shelters in the UK, including a comparison with bio-based shelters made from corn-derived starch and PLA
19	Chen et al. (2016)	P/A	Comparison of traditional petroleum-based PET bottles and partially or fully bio-based PET bottles from different feedstocks (corn, switchgrass or wheat straw for ethylene glycol; corn stover or forest residues for terephthalic acid) in the US
20	Deng et al. (2013)	P/A	Wheat-gluten-based powder and wheat-gluten-based packaging film production compared to corn-based PLA and LDPE packaging films in Europe

<b>N.</b>	<b>Reference</b>	<b>Product/study category (1)</b>	<b>Scope / Description</b>
21	Razza et al. (2015)	P/A	Bio-based and biodegradable foamed packaging for protecting washing machines made of starch from tapioca, potato and corn, and comparison with conventional fossil-based EPS (Expanded Polystyrene) cushioning packaging under European conditions
22	van der Harst et al. (2014)	P/A	Comparison of disposable beverage cups made of PS, PLA and paperboard coated with bioplastic (PLA) in the Netherlands
23	Bisinella et al. (2018)	P/A	Production, use and disposal of grocery carrier bags of different types (disposable/reusable) and materials including LDPE, r-LDPE, PP, r-PET, PET-based polyester, starch-polyester biopolymer, paper, cotton and composite (jute, PP, cotton), in Denmark
24	Detzel and Krüger (2006)	P/A	Comparison of clam shells made from corn-based PLA, PET, PP and oriented PS in Germany
25	Gérand and Roux (2014)	P/A	Partly recycled PET bottle (75 cl) compared to glass bottle for wine in France, including the life cycle of packaged wine
26	Parker and Edwards (2012)	P/A	Comparison of a conventional HDPE bag, an oxo-biodegradable HDPE bag and a bio-based (starch-polyester) bag for use as carrier bags and bread packaging in the UK
27	Markwardt et al. (2017)	P/A	Comparison of Tetra Pak carton packages and alternative packages (based on PET, HDPE and glass) for different beverage sectors in the European Nordic markets. Compared packaging alternatives also include bio-based HDPE from sugarcane-derived ethanol and partially recycled PET
28	Müller (2012)	P/A	Comparison of compostable ( <i>ecovio</i> <sup>®</sup> )( <sup>2</sup> ) bags with PE (Polyethylene) and paper bags for transportation of staple goods, and disposal of organic waste after reuse for other purposes in Germany
29	Müller and Müller (2017)	P/A	Comparison of fruit and vegetable bags made of different materials (HDPE, paper, compostable <i>ecovio</i> <sup>®</sup> polymer( <sup>2</sup> )) in France
30	Müller and Müller (2015)	P/A	Comparison of the use of biodegradable ( <i>ecovio</i> <sup>®</sup> ) ( <sup>2</sup> ) and conventional PE mulch film for cotton growing in China

N.	Reference	Product/study category <sup>(1)</sup>	Scope / Description
31	Guo et al. (2013)	EoL	Comparison of alternative waste management options (landfill, anaerobic digestion, industrial composting and home composting) for a display board made of starch-PVOH (Polyvinyl alcohol) biopolymer foam from three different feedstocks (wheat, potato and maize) in the UK
32	Rossi et al. (2015)	EoL	Comparison of end-of-life options for unspecified PLA and TPS packaging for dry products, derived from corn (mechanical recycling, industrial composting, anaerobic digestion, direct fuel substitution in industrial facilities, municipal incineration with heat recovery, landfilling)

<sup>(1)</sup> The following abbreviations are used: M/I (monomer/intermediate), P (polymer), P/A (plastic product/article), EoL (End-of-Life related study).

<sup>(2)</sup> *ecovio*® is the commercial name for a compostable polymer blend made of PLA (Polylactic Acid) and PBAT (Polybutylene Adipate co-Terephthalate; a co-polyester of 1,4-butanediol, adipic acid and terephthalic acid).

#### **K.1.4. In-depth assessment**

In the in-depth assessment, information was collected about the methodological and modelling choices performed in each study with respect to aspects considered relevant in relation to LCA of plastics and plastic products. The focus was on most challenging and/or controversial aspects.

Regarding feedstock supply, material production and product manufacturing, the aspects addresses in the in-depth assessment include:

- Modelling of agricultural production processes (with a special focus on fertiliser and pesticide emissions);
- Inclusion of direct and indirect land use changes (dLUC/iLUC) and related modelling approach;
- Range of emissions covered in the modelling of dLUC and iLUC (e.g. only GHGs or also other emissions);
- Handling of biogenic carbon emissions and removals from the product, i.e.:
  - Inclusion of a specific biogenic carbon balance/inventory,
  - Modelling of biogenic carbon emissions/removals from the product at the inventory level,
  - Accounting of any form of “temporary” or “permanent” biogenic carbon storage (from the modelled emissions and removals), and
  - Inclusion of any form of dynamic accounting/assessment of (delayed) carbon emissions and removals;
- Modelling of the use of waste or residual bio-based feedstocks;
- Handling of multi-functionality at the biorefinery level (due to co-production of multiple outputs);
- Inclusion of plastic additives and related modelling and impacts.

Regarding End of Life modelling, the following aspects were instead considered:

- Modelling of product/material biodegradation during biological treatment or in-situ degradation (i.e., applied degradation rates, overall modelling approach, etc.);
- Accounting of potential consequences of using biodegradable plastics on organic waste management (e.g. reduction of impurities, increased separate collection rate, etc.) or other potential indirect effects;
- Inclusion of littering generation, release and impacts (and related modelling approach);
- Inclusion of micro-plastics generation, release and impacts (and related modelling approach).

Finally, general methodological aspects were addressed, including:

- LCA software used (if any);
- Database used for secondary/background data;
- Impact assessment level (midpoint or endpoint);
- Considered impact categories;
- Impact assessment method.

## **K.2. Results**

This section summarises the main findings of the systematic review. An overview of the findings related to the most relevant general aspects analysed in the screening assessment is presented first, which are mostly related to the scope of the collected studies. This is followed by a description of the results related to the most relevant methodological and modelling aspects considered in the in-depth assessment.

### **K.2.1. Most relevant general aspects (screening assessment - all collected studies)**

The main general aspects considered in the screening assessment that are addressed in this result section are: (a) the products analysed in the studies, (b) the related types of feedstock, and (c) the applied system boundary. Results related to the other considered general aspects are not addressed here, as they were mainly used to select a restricted set of relevant studies for the in-depth assessment, and considering the primary focus of the review on applied methodological and modelling choices.

#### **K.2.1.1. Products**

A huge variety of products was addressed in the 171 screened studies, also due to the breadth of the applied search criteria during the collection of potentially relevant studies for the review (Section K.1.1). Most of the studies focusing on monomers and intermediates referred to a specific and normally different substance. Only a few monomers were addressed in more than one study, i.e. ethylene (3 studies), succinic acid (3 studies), lactic acid (2 studies), and propylene glycol (2 studies). Among polymers based on alternative feedstocks, the most considered ones were Poly-Lactic Acid (PLA) (14 studies), followed by Poly-Hydroxy-Alkanoates (PHAs) and Poly-Hydroxy-Butyrates (PHBs), which were addressed in 10 studies (i.e. 7 on PHAs and 3 on PHBs). The studies on plastic products/articles mostly focused on PLA or PLA-based products (e.g. bottles, cups, packaging, etc.; 20 studies) highlighting the relevance of this polymer for different final applications. Five studies specifically referred to products based on Ecovio® or Ecoflex® as material (e.g. mulching film). Overall, 111 out of 171 studies focused on biodegradable products under different controlled conditions (e.g. industrial composting) or in-situ, while 49 studies addressed non-biodegradable products (in the remaining 11 studies, no explicit mention regarding biodegradability of the product was reported). On the other hand, the entire set of end-of-life related studies (14 in total) focused on biodegradable products.

#### **K.2.1.2. Feedstocks**

Regarding the feedstock, maize (in the form of grain, stover, or respective derivatives) was the most frequently considered alternative feedstock in the screened studies. This also reflects the relative abundance of studies focusing on Polylactic Acid (PLA) as a polymer or on PLA/PLA-based products (Section K.2.1.1), which is currently often derived from starch.

#### **K.2.1.3. System boundary**

Focusing on the system boundary applied in the screened studies, the overall picture is shown in Figure K.1 and Table K.2. Five main categories of studies were detected with respect to the applied system boundary: (i) full "*Cradle-to-Grave*" studies covering the entire product life cycle<sup>180</sup> (albeit possibly excluding one or more intermediate stages or activities considered not relevant for the assessment, such as the Use stage and/or retail activities occurring during the Distribution stage); (ii) "*Cradle-to-Gate + Grave*" studies, which compared to full cradle-to-grave studies excluded intermediate stages occurring between Product Manufacturing and End of Life; (iii) "*Cradle-to-Gate*" studies limiting the

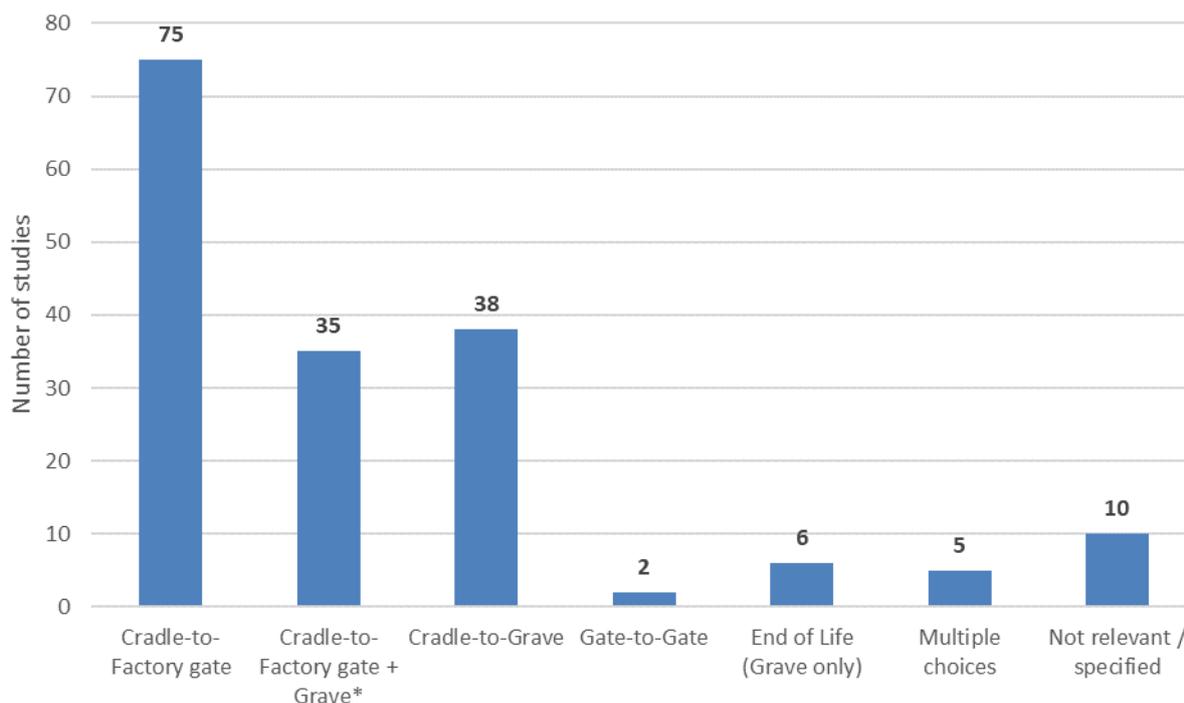
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<sup>180</sup> Note that for studies investigating waste or residual feedstocks the "cradle" considered in the assessment may not correspond with resource extraction from the environment or biomass production.

system boundary to those stages occurring from Raw Material Acquisition to Manufacturing of the product (albeit including in some cases its Distribution or at least transport until the final user); (iv) "Gate-to-Gate" studies focusing only on one specific intermediate stage or process chain (e.g. polymer production); and (v) end-of-life related studies focusing only on the End of Life stage (i.e. the "grave"). A few studies conducting separate assessments with different system boundaries for the same product(s) were also detected, and accounted separately (e.g. studies combining both a Cradle-to-Gate and a Cradle-to-Grave perspective). This was also the case of studies applying different system boundaries for different products or feedstock use scenarios.

In the vast majority of the studies dealing with monomers and intermediates (i.e. 30 out of 34), the system boundary was defined as Cradle-to-Gate. However, three studies also considered the Use stage or the End of Life stage, either as a unique choice of the assessment (Tao et al., 2014), in a separate additional evaluation (Adom and Dunn, 2017), or depending on the final use considered for the feedstock (Alvarenga and Dewulf, 2013c). In particular, Adom and Dunn (2017) calculated Cradle-to-Gate results for Polymer-Grade Lactic Acid (PGLA) and ethyl lactate, but additionally investigated biogenic carbon emissions in a Cradle-to-Grave perspective (based on assumptions for the End of Life of an unspecified product derived from these monomers, and the associated fate of carbon). Similarly, Alvarenga and Dewulf (2013c) adopted a Cradle-to-Gate perspective when assessing the use of Brazilian bioethanol for bio-based ethylene production as an alternative to fossil-based feedstock used for conventional ethylene production. However, the system boundary was extended until bioethanol use in vehicles when evaluating this scenario as an alternative to gasoline use. Conversely, Tao et al. (2014) compared cellulosic iso-butanol with cellulosic ethanol and n-butanol directly under a Cradle-to-Grave perspective (i.e. considering their use for fuel production and subsequently in vehicles).

Also the studies focusing on polymers mostly adopted a Cradle-to-Gate approach (35 out of 58 studies). Nevertheless, eleven studies included also the End of Life stage in the system boundary, while still excluding the Use stage and other intermediate stages. Moreover, two studies applied a full Cradle-to-Grave approach, i.e. Renouf et al. (2013) to Poly-Lactic Acid (PLA) from sugarcane (when comparing different production, processing and use pathways for this feedstock), and Roes and Patel (2007) to evaluate life cycle risks for human health associated with Poly-Trimethylene Terephthalate (PTT), Poly-Hydroxy-Alkanoates (PHAs), Polyethylene Terephthalate (PET), and Polyethylene (PE) from bio-based and petrochemical feedstocks. On the other hand, two studies restricted the system boundary to only one stage or process chain, according to a Gate-to-Gate approach, to either evaluate the potential environmental impacts of a novel protocol for the extraction of Poly-Hydroxy-Butyrate (PHB) based on dimethyl carbonate (DMC) (Righi et al., 2017), or to assess the energy demand of PLA, PHBV (PolyHydroxyButyrate-co-Valerate), Bio-PE and PP (Polypropylene) manufacturing in three different processes (Schulze et al., 2017). Similarly, a couple of studies adopted an End-of-Life focused perspective, to evaluate alternative treatment and use options for wastewater, including the integration of mixed culture PHA production (Morgan-Sagastume et al., 2016; municipal wastewater) or its application as an alternative to biogas production (Gurieff and Lant, 2007; industrial wastewater from food industry). The objectives was hence not to explicitly investigate the potential impacts associated with PHA production. Finally, Hansen et al. (2015) evaluated Cradle-to-Gate impacts of bio-based and fossil-based GP-PS and HI-PS (General Purpose and High Impact Polystyrene) production in Brazil, but additionally calculated the overall carbon balance of the different investigated scenarios also considering a hypothetical End of Life stage for the polymers (assumed to be incineration).



Notes: General - Only Cradle-to-Grave studies are foreseen in the context of ISO 14040/44 standards; (\*) The Use stage and other intermediate stages are not included in the system boundary.

**Figure K.1.** System boundary applied in the screened LCA studies, regardless of the specific product or study category.

The analysis of the studies dealing with finished plastic products and articles highlighted the importance of taking the End of Life stage into account in the assessment. Indeed, approximately 80% of the studies (51 over 65) set the system boundary according to a Cradle-to-Grave approach (33 studies, in a few cases excluding the Use and/or retail stages), or as “Cradle-to-Gate plus Grave” boundary (i.e. including the End of Life stage, but not the Use of the product/article or other intermediate stages; 18 studies). Only ten studies defined the system boundary as Cradle-to-Gate, although three of them also considered the distribution/transport of the plastic product or article to the consumer as part of the system (i.e. ultimately terminating the boundary at the consumer gate). In one study, a Cradle-to-Grave assessment of packaging film made of different alternative materials followed a more restricted, Cradle-to-Gate evaluation of the potential impacts of one of such materials (Deng et al., 2013).

Four of the fourteen studies specifically comparing alternative end-of-life options for a number of plastic products and materials considered only the End of Life stage (i.e. the “grave”). Most of the remaining studies (9 out of 14) conducted this end-of-life comparison also considering upstream life cycle stages, either following a full Cradle-to-Grave perspective (3 studies), or still excluding the Use stage and any other intermediate stage (“Cradle-to-Gate plus Grave” approach; 6 studies). On the other hand, one study complemented this end-of-life assessment with a comparison of cradle-to-gate impacts of the investigated products (i.e. PLA and PET bottles or starch-based and PE bags; Gironi and Piemonte, 2010).

**Table K.2.** System boundary applied in the screened LCA studies, subdivided per product or study category <sup>(1)</sup>.

System boundary <sup>(2)</sup>	Product / study category				
	Monomers/ Intermediates	Polymers	Products or articles	EoL-related studies	Total
<i>C-FG</i>	30	35	10	-	75
<i>C-FG + GR</i>	0	11	18	6	35
<i>C-GR</i>	1	2	33	2	38
<i>G-G</i>	-	2	-	-	2
<i>EoL (only "grave")</i>	-	2	-	4	6
<i>Multiple system boundaries</i>	2 <sup>(3)</sup>	1 <sup>(3)</sup>	1	1 <sup>(4)</sup>	5
<i>N.a. <sup>(5)</sup></i>	1	5	3	1	10
<b>TOTAL</b>	<b>34</b>	<b>58</b>	<b>65</b>	<b>14</b>	<b>171</b>

<sup>(1)</sup> Note that only Cradle-to-Grave studies are foreseen in the context of ISO 14040/14044 standards.

<sup>(2)</sup> Acronyms: C-FG = Cradle-to-Factory Gate; C-FG+GR = Cradle-to-Factory Gate + Grave (the Use stage and other intermediate stages are not included); C-GR = Cradle-to-Grave; EoL = only the End of Life stage is included; G-G = Gate-to-Gate.

<sup>(3)</sup> Including Cradle-to-Gate studies complementing the assessment with a Cradle-to-Grave quantification of the product carbon balance, or studies applying both perspectives (Cradle-to-Gate and Cradle-to-Grave) depending on the production, processing or use scenario considered for the feedstock.

<sup>(4)</sup> Study complementing the end-of-life focused evaluation with a Cradle-to-Gate perspective.

<sup>(5)</sup> "N.a." includes reviews, non-LCA studies, and those studies where the system boundary was not specified.

### K.2.2. Methodological and modelling aspects (in-depth assessment)

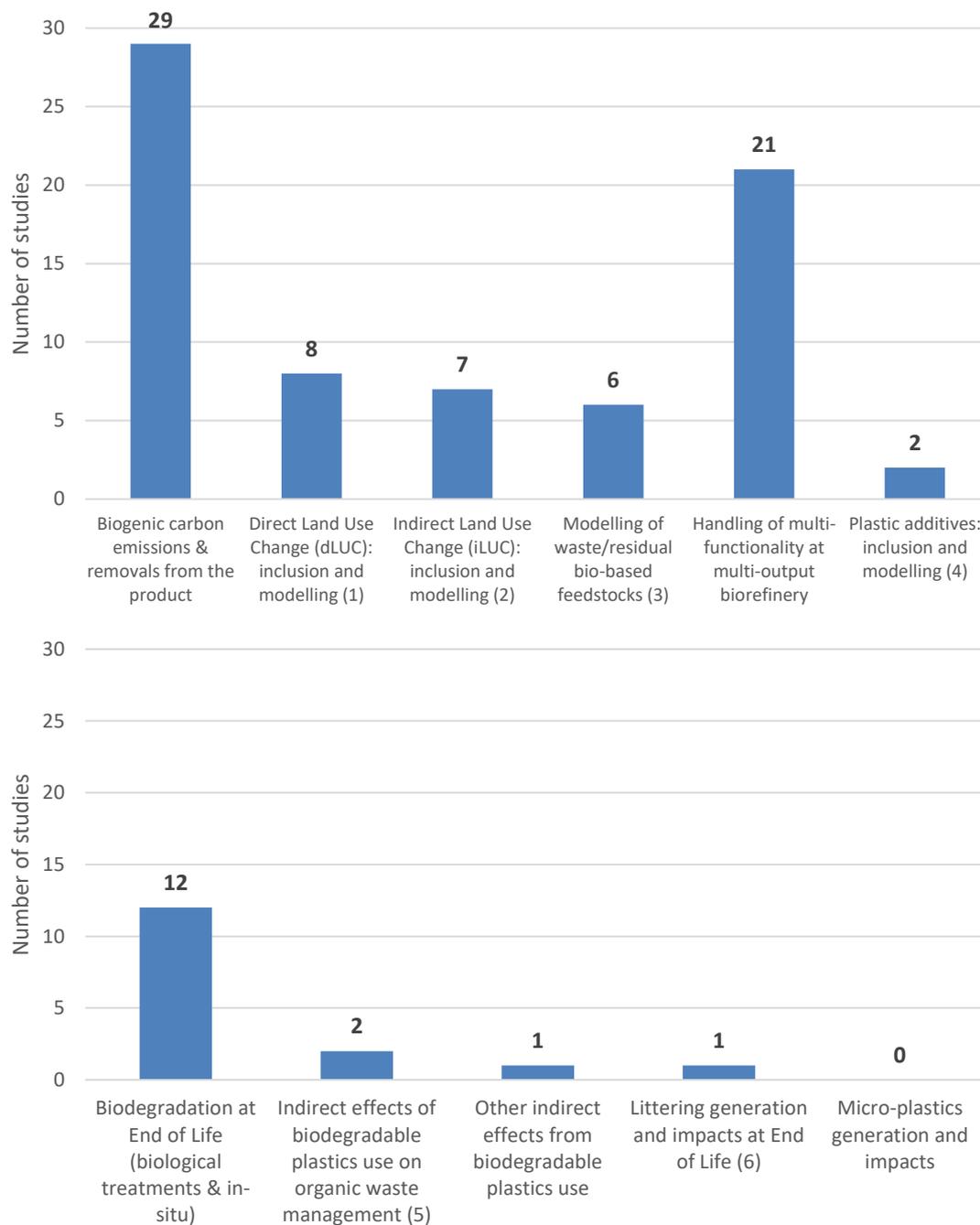
This section presents the main findings of the review related to the methodological and modelling aspects addressed in the in-depth assessment. Unless stated otherwise, the presented results refer to the 32 studies specifically selected for such detailed analysis. However, for some methodological aspects that were addressed only a limited number of selected studies (i.e. direct and indirect land use changes and modelling of waste/residual bio-based feedstocks), relevant information was retrieved also from studies that were not selected for the in-depth assessment, but only included in the initial screening phase. This is clearly specified throughout the next sections, and was made to achieve a sufficiently comprehensive picture of applied methodological choices with respect to such aspects, and to avoid disregarding relevant information.

To provide an initial overview, Figure K.2 presents the total number of selected studies where the different methodological/modelling aspects considered in the in-depth assessment were addresses explicitly and with a sufficient level of detail to understand the applied methodological or modelling choices. However, apart from some exceptions<sup>181</sup>, such figures shall not be interpreted as an indicator of whether specific aspects are more or less widely and/or appropriately addressed in the current literature

<sup>181</sup> Related to aspects that are relevant to all plastic products (regardless of the feedstock applied), i.e. inclusion and modelling of plastic additives, as well as littering and micro-plastics generation and impacts (although additives and littering are not relevant to the few –i.e. 4- studies focusing on intermediates).

(as represented in the sample of reviewed studies), as the inclusion of a given aspect in a study also depends on its relevance for the product(s) in scope, and not necessarily on the deliberate choice of including or excluding it. For instance, modelling of waste/residual bio-based feedstocks is not relevant if these are not used for the product in scope, while inclusion and modelling of land use changes is equally not relevant if no land-based products are investigated in the study (although even in this case they may not be necessarily addressed).

In general terms, the most frequently addressed methodological aspect was handling of biogenic carbon emissions and removals related to the biogenic carbon content in products (29 studies). This is followed by handling of multi-functionality at the biorefinery level (due to production of multiple outputs; 21 studies) and, to a lower extent, by modelling of biodegradation during biological treatments or in-situ at End of Life (12 studies). Direct and indirect land use changes and modelling of waste/residual bio-based feedstocks were addressed only in a limited number of studies (from 6 to 8, depending on the issue), although even several additional studies excluded from the in-depth assessment also addressed such aspects (especially the use of waste/residual bio-based feedstocks; 21 studies). Conversely, plastics additives and related modelling were addressed only in two studies excluded from the in-depth assessment (if a couple of additional studies focusing on the use of additives to improve a specific product performance, and equally not analysed in detail, are not taken into account). Similarly, potential indirect effects from using biodegradable plastics (i.e. carrier bags and cutlery/tableware) on organic waste management were considered in two studies, other than two additional studies not investigated in detail. Littering generation at End of Life and related potential impacts were at least partially addressed only in one study, building upon preliminary approaches discussed in two previous studies not included in the in-depth assessment (and outside the timeframe considered in this review). On the other hand, micro-plastics generation, release and potential impacts were not addressed in any of the investigated studies, nor in those collected for possible review. Detailed results for the single methodological aspects addressed in the in-depth assessment are presented in the following subsections (K.2.2.1–K.2.2.10).



Notes: (1) At least 5 studies excluded from in-depth assessment also addressed dLUC; (2) Also 9 studies excluded from in-depth assessment addressed iLUC; (3) Also 21 studies excluded from in-depth assessment addressed the use of waste/residual bio-based feedstocks; (4) Referring to studies not included in the in-depth assessment and excluding those considering the use of additives to improve a specific product performance (e.g. shelf life of food); (5) At least 2 studies excluded from in-depth assessment also addressed indirect effects of biodegradable plastics on organic waste management; (6) Also 2 studies excluded from in-depth assessment (and outside the timeframe considered for this review) partially addressed littering generation and impacts.

**Figure K.2.** Number of studies selected for in-depth assessment (unless otherwise stated) addressing explicitly and with sufficient level of detail the different methodological aspects investigated in the review (when relevant for the product(s) in scope).

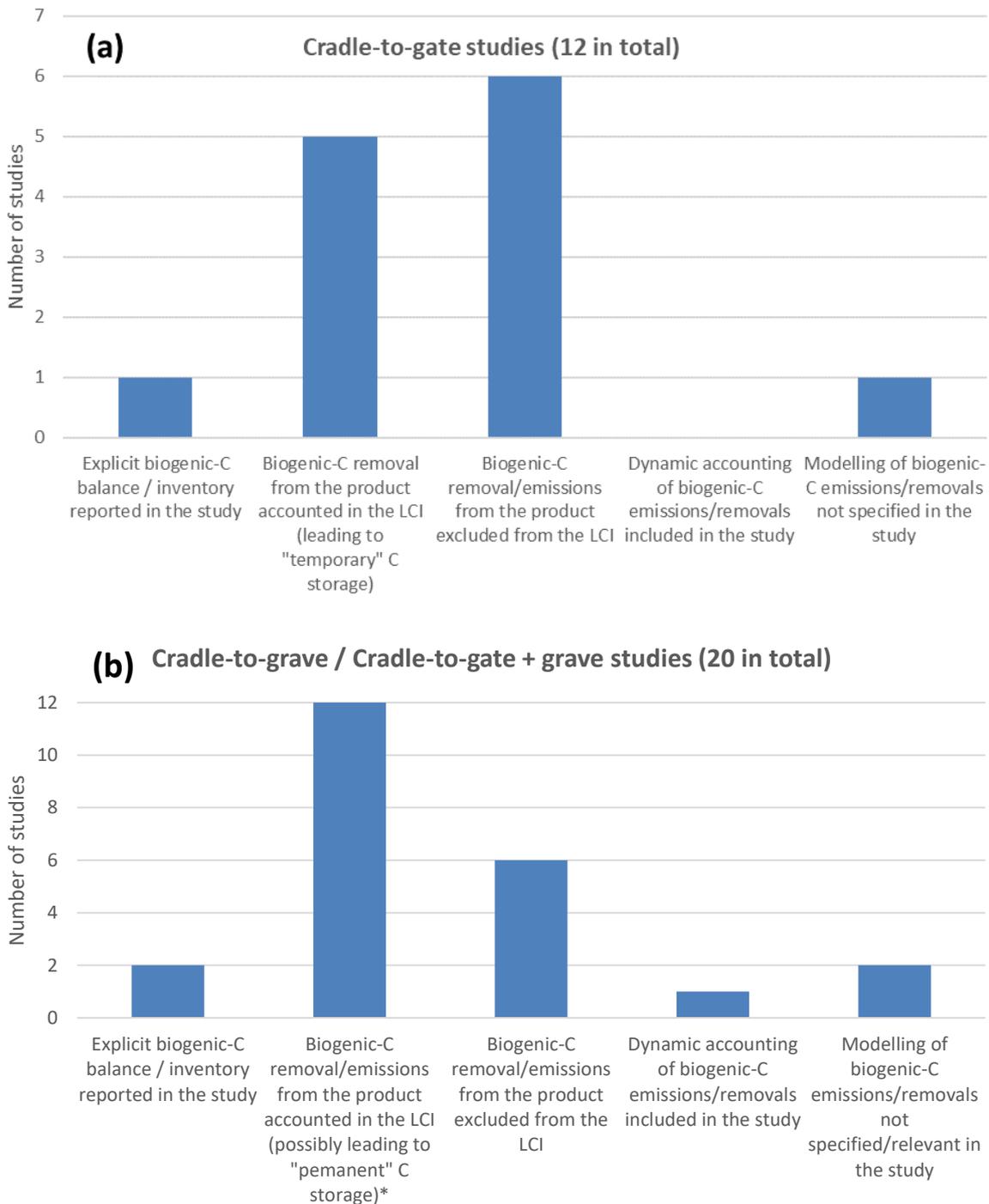
### ***K.2.2.1. Handling of biogenic carbon emissions and removals from the product***

Handling of biogenic carbon (especially CO<sub>2</sub>) emissions and removals from products represented a relevant methodological aspect for almost the entire set of studies included in the in-depth analysis, as all of them except one investigated at least one bio-based product (either final or intermediate; i.e. in 31 studies out of 32 included in the in-depth analysis). However, such aspect was addressed to different extents and with different levels of detail across the single studies. In some cases, applied choices were also not explicitly described, and wherever possible they had to be inferred from the results and from any other useful element or description reported throughout the study. In a few other cases, no sufficient information was available for this purpose. The main findings of the review for this methodological issue are summarised below, while a general overview is presented in Figure K.3. A distinction is made between partial, cradle-to-gate studies of intermediate or final products, and full cradle-to-grave studies of final products (or anyway cradle-to-gate studies including also the End of Life stage of either final or intermediate products). This is because not all aspects related to the modelling of biogenic carbon emissions and removals are relevant for both categories of study (e.g. dynamic accounting of carbon emissions is generally not relevant for cradle-to-gate studies), while other choices are inherently affected by the applied system boundary (e.g. release at End of Life is typically not modelled in bare cradle-to-gate studies). Investigated aspects were the inclusion of an explicit biogenic carbon balance/inventory in the study, the specific modelling of biogenic carbon emissions and removals from the product at the inventory level, the accounting of any subsequent form of “temporary” or “permanent” carbon storage, and the possible inclusion of any additional type of dynamic accounting of carbon emissions and removals.

None of the reviewed cradle-to-gate studies (12 out of 31 addressing at least one bio-based feedstock) included an explicit, dedicated balance or inventory of biogenic carbon emissions and removals from the product, although this reporting may be considered less relevant when downstream life cycle stages are not addressed. However, Hansen et al. (2015), included a detailed balance when additionally considering an expanded systemic perspective including also the product End of Life (assumed to be incineration), to explore the effects of accounting for all possible forms of carbon emissions and removals on the overall balance. In half (six) of the cradle-to-gate studies biogenic carbon emissions and removals were not modelled in the inventory, likely to avoid the calculation of potentially distorted results due to the exclusion of emissions from End of Life (the applied system boundary ends at the factory gate). In particular, Forte et al. (2016) reported that no CO<sub>2</sub> uptake was modelled as the long-term storage of carbon is highly dependent on the final use of the investigated biopolymer (which was unknown) and durability of the corresponding application. On the other hand, five studies accounted for the removal of biogenic carbon (CO<sub>2</sub>) incorporated in the product, either modelling the corresponding uptake during biomass cultivation (4 studies), or by crediting the biogenic carbon content in the product as a negative emission to the product system (1 study; Tsiropoulos et al., 2015). In both cases, the inventory consequently acknowledged that carbon taken up from the atmosphere is temporarily “stored” in the product, until it will be released back during Use or End of Life. However, this aspect was not necessarily communicated correctly across the results of the different studies. Only in one study the applied modelling approach was not specified (Daful et al., 2016). Due to the cut-off at the factory gate, dynamic accounting of carbon emissions and removals (considering the timing of uptakes and releases over a fixed time horizon) was not implemented in any of the investigated cradle-to-gate studies.

Focusing on studies accounting also for the End of Life stage (i.e. full cradle-to-grave or cradle-to-gate plus grave studies; 19 out of 31 addressing at least one bio-based feedstock), an explicit balance/inventory of biogenic carbon emissions and removals from the product was included only in a couple of them (i.e. Razza et al., 2015 and Rossi et al., 2015). As for the modelling of these emissions and removals, almost two thirds of the mentioned studies (i.e. 12 out of 19) included the removal of biogenic carbon incorporated in the product, mostly considering the corresponding uptake during

biomass cultivation (8 studies), or less frequently by crediting the biogenic carbon content of the product as a negative emission to the product system (3 studies). Subsequent release at End of Life was then also modelled in such studies, although in several cases the way this release was exactly considered to occur from the single end-of-life options applied to the product was not (clearly) specified. In one study, only the net amount of biogenic carbon in the product assumed to be no longer released at End of Life (after landfilling or on-land application of composted material) was apparently modelled, as a negative emission from the specific End of Life activity (Razza et al., 2015). However, most of these studies did not clearly describe the existence of any time horizon after which emissions from End of Life were no longer modelled (and hence any remaining, non-released carbon was considered as “permanently” stored in the product). For instance, in the case of landfilling, a time horizon of 100 years was explicitly or implicitly applied in some studies, and biogenic carbon not degraded during such period was considered to be no longer released back to the atmosphere (e.g. Razza et al., 2015; Posen et al., 2016; Hottle et al., 2017). In these and other studies, non-degraded biogenic carbon after composting was also (apparently) assumed to be no longer released to the atmosphere, although the applied time horizon was generally not reported, and it was not always clear whether subsequent emissions from compost use (e.g. on-land application) were properly taken into account in the modelling (e.g. in Detzel and Krüger, 2006, beyond the studies mentioned above). Rossi et al. (2015) consistently applied a 100-year time horizon throughout all the modelled end-of-life options (including mechanical recycling, industrial composting, anaerobic digestion, direct fuel substitution, municipal incineration and landfilling). Similarly, emissions from incineration were also generally modelled consistently across the different studies, and fully attributed to the product in scope. Conversely, Van der Harst et al. (2014) assumed emissions at End of Life to be identical to the uptake during feedstock cultivation, considering a short carbon cycle for disposable beverage cups. In the remaining studies analysed in the review, biogenic carbon emissions and removals were either not modelled in the inventory (6 studies), or no sufficiently detailed information was provided in this respect (2 studies, with one of them not addressing any bio-based feedstock). Biogenic carbon emissions were modelled and accounted dynamically only in one study (Rossi et al., 2015), considering their timing over a 100-year time horizon. Biogenic carbon removals from biomass were instead not timed in such study, as they were considered to take place over a relatively short period (less than one year) for crops used as feedstock for the investigated bio-based polymers (i.e. potato and maize). Finally, while not specifically applying any dynamic accounting approach, Kendall et al. (2012) conducted a scenario analysis considering different time horizons (i.e. 20, 50 and 100 years) to account for carbon emissions from landfill gas generation.



(\*) Depending on assumed release and (bio)-degradation pathways at End of Life, and on application of any time horizon after which emissions are no longer modelled (both often not clearly specified across the reviewed studies).

**Figure K.3.** Handling of different aspects related to the modelling of biogenic carbon emissions and removals in the reviewed studies, differentiating between cradle-to-gate studies of intermediate/final products (a) and cradle-to-gate / cradle-to-gate plus grave studies of final or intermediate products also including End of Life (b).

### **K.2.2.2. Direct Land Use Change**

Direct land use change (dLUC) was explicitly accounted in eight out of the 32 studies included in the in-depth analysis<sup>182</sup>, as reported in Table K.3 (Liptow and Tillman, 2012; Alvarenga et al., 2013a,b; Hansen et al., 2015; Razza et al., 2015; Tsiropoulos et al., 2015; Posen et al., 2016; Markwardt et al., 2017). Only GHG emissions potentially associated to dLUC were taken into account in these studies, and quantified based on the models, approaches or emission factors described below and summarised in Table K.3. The emissions from dLUC were either quantified independently, or as part of overall land use change emissions incorporating explicitly or implicitly also the contribution of iLUC.

In Liptow and Tillman (2012), dLUC was considered only when a consequential approach was applied and expansion of sugarcane cultivation to virgin areas of Brazil was assumed to take place. In this case, a release of 1040 kg CO<sub>2</sub> eq. per m<sup>3</sup> of ethanol was assumed, based on literature (Zuurbier and Van de Vooren, 2008). Alvarenga et al. (2013a) modelled dLUC based on historical literature data on the type of land affected by the expansion of sugarcane cultivation in the state of Sao Paulo (Brazil) between 2003 and 2009 (taken from Rudorff et al., 2010), considering that dLUC occurred only in 15.5% of the total sugarcane area as of 2009 (the rest being areas already cultivated before 2003). Total estimated GHG emissions from dLUC were distributed over 20 years. In contrast, Alvarenga et al. (2013b), who apply a consequential approach, considered that sugarcane cultivation takes place only on newly converted land, which was previously pasture land, since this is the assumed trend for the future. In both studies by Alvarenga and colleagues, GHG emission factors for the assumed land conversions were estimated based on the approach recommended in the ILCD Handbook (EC-JRC, 2010), and accounted for changes in soil organic carbon content. Hansen et al. (2015) calculated carbon emissions from dLUC (land transformation) based on the difference between soil carbon stocks before and after the installation of the sugarcane crop, using the IPCC (2006) Tier 1 approach, in accordance with the EC guidelines for the calculation of land carbon stocks (EC, 2010) complementing the Renewable Energy Directive (RED) 2009/28/EC. Tsiropoulos et al. (2015) included overall GHG emissions due to LUC (both direct and indirect) from sugarcane cultivation, based on the upper and lower bound of the range of emission factors provided in the review of (indirect) land use change models by Wicke et al. (2012; e.g. 3-46 g CO<sub>2</sub> eq./MJ of ethanol). Posen et al. (2016) applied the Carbon Calculator tool for Land Use Change (CCLUB) included in the GREET model (Wang, 2014), which is based on a general economic equilibrium model to predict global land use changes (seemingly both direct and indirect) from biofuel production (i.e. the GTAP model), combined with other agroecosystem and crop models (e.g. the CENTURY model). The authors assumed that the contribution of bio-based plastics to land use change would be similar to that of biofuel production (per unit of feedstock diverted to bioplastic/biofuel production). A distribution of overall LUC GHG emissions from land use changes predicted applying the models was estimated in the study, based on a combination of the results from the main scenarios implemented in the CCLUB tool. Razza et al. (2015) assessed the impact of dLUC in a sensitivity analysis considering both a best-case and a worst-case scenario, where previous land use was either grassland (best case) or rainforest (worst case). GHG emission factors were taken from PAS 2050 (BSI, 2011), and factors for Malaysia were used as proxy for emissions occurring in Thailand. Finally, Markwardt et al. (2017) performed a sensitivity analysis for a beverage packaging applying a LCI for bio-based PE developed by Braskem, where the effect of dLUC is addressed (Ziem et al., 2013)<sup>183</sup>. For dLUC, both a current and a future scenario was modelled by Ziem and colleagues, based on land types converted in the two preceding seasons in the specific regions where the feedstock is sourced, or those types that are expected to be converted in the near future, respectively. The

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<sup>182</sup> Of these, 29 studies have primary bio-based feedstocks that may generate dLUC in scope.

<sup>183</sup> Note that this study has also been collected and screened for review. While it was ultimately not included in the in-depth assessment, the approach applied to address dLUC is briefly discussed below, for completeness.

corresponding GHG emissions were estimated considering the EC guidelines for the calculation of land carbon stocks for the purpose of the RED (EC, 2010).

At least five studies not included in the in-depth analysis also addressed dLUC (alone or as part of overall LUC emissions), at least in a sensitivity analysis (Gironi and Piemonte, 2010; Ekman and Börjesson, 2011; Piemonte and Gironi, 2011; Piemonte and Gironi, 2012; Ziem et al., 2013). Gironi and Piemonte (2010) accounted for the overall land use change emissions from land demanded for bioplastic production applying a unique GHG emission factor calculated as the average of the values reported by Searchinger et al. (2008; 351 t CO<sub>2</sub> eq/ha of land converted to cropland), and Righelato and Spracklen (2007; 305 t CO<sub>2</sub> eq/ha of land converted). The resulting average value (328 t CO<sub>2</sub> eq/ha of land converted) was consistently applied also in Piemonte and Gironi (2011) and Piemonte and Gironi (2012). Ekman and Börjesson (2011) considered dLUC in a sensitivity analysis taking into account the additional land area needed to supply sugar for propionic acid production. Under current Swedish conditions, the land area was assumed to be taken from previous grassland leading to additional biogenic CO<sub>2</sub> emissions from the soil, equivalent to 350–450 kg C/ha and year (based on Börjesson and Tufvesson, 2011, Cherubini et al., 2009a, and Cherubini et al., 2009b). Finally, as already mentioned above, Ziem et al. (2013; which was also independently investigated in the screening assessment) quantified dLUC emissions from the assumed land conversions according to the EC guidelines for the calculation of land carbon stocks within the context of the RED (EC, 2010).

**Table K.3.** Models, guidelines or emission factors applied in the reviewed studies to quantify GHG emissions from dLUC.

<b>Model / Guideline / Emission factor (source)</b>	<b>Number of studies</b>	<b>Reference(s)</b>
EC guidelines for the calculation of land carbon stocks (EC, 2010) + IPCC (2006) (where appropriate)	3 (2 from in-depth assessment; 1 from screening assessment)	Hansen et al. (2015) Markwardt et al. (2017) <i>Ziem et al. (2013) <sup>(1)</sup></i>
Guidelines from the ILCD Handbook, accounting for soil organic carbon stock changes (EC-JRC, 2010)	2	Alvarenga et al. (2013a) Alvarenga et al. (2013b)
Emission factors from PAS 2050 (BSI, 2011)	1	Razza et al. (2015)
CCLUB tool for LUC included in the GREET model (Wang, 2014)	1	Posen et al. (2016)
Emission factors from literature (Righelato and Spracklen, 2007; Searchinger et al., 2008; Zuurbier and Van de Vooren, 2008; Cherubini et al., 2009a and 2009b; Börjesson and Tufvesson, 2011; Wicke et al., 2012)	6 (2 from in-depth assessment; 4 from screening assessment)	Liptow and Tillman (2012) Tsiropoulos et al. (2015) <i>Gironi and Piemonte (2010) <sup>(1)</sup></i> <i>Ekman and Börjesson (2011) <sup>(1)</sup></i> <i>Piemonte and Gironi (2011) <sup>(1)</sup></i> <i>Piemonte and Gironi 2012) <sup>(1)</sup></i>

<sup>(1)</sup> Study not included in the in-depth assessment. Due to the limited number of selected studies addressing dLUC, this overview table has been extended also to studies that otherwise were not assessed in-depth (as only included in the screening assessment).

### **K.2.2.3. Indirect Land Use Change**

Indirect land use change (iLUC) was addressed in 7 out of the 32 studies included in the in-depth analysis<sup>184</sup>, as detailed in Table K.4 (Kendall et al., 2012; Liptow and Tillman, 2012; Shen et al., 2012; Alvarenga et al., 2013b; Tsiropoulos et al., 2015; Posen et al.; 2016; Parajuli et al., 2017). In all these studies, only GHG emissions potentially arising from iLUC were considered, and quantified according to the models, approaches or emission factors described below and summarised in Table K.4. Emissions were either quantified independently, or as part of overall land use change emissions incorporating explicitly or implicitly also the contribution of dLUC.

Kendall et al. (2012) applied an iLUC GHG emission factor derived from earlier applications of the Argonne National Lab's GREET model (Wang, 2009), equalling 15.97 g CO<sub>2</sub>eq/kg glucose. This is, however, the result of older studies relying on GREET. Liptow and Tillman (2012) used two extreme emission factors to calculate the iLUC impact on climate, considering both a "low impact value" of zero (i.e. no impact) and a "high impact value" equal to 46 g CO<sub>2</sub>/MJ sugarcane-derived ethanol (taken from California EPA, 2009). Shen et al. (2012) applied standard emission factors extracted from the same source, which at the time was the first one providing government-adopted iLUC emission factors (30 g CO<sub>2</sub> eq./MJ maize-based ethanol and 46 g CO<sub>2</sub> eq./MJ sugarcane-based ethanol). In Alvarenga et al. (2013b), the calculation was based on the assumption that pasture lands displaced by sugarcane expansion (as a result of dLUC) would move to areas with natural vegetation, i.e. the Amazon Forest. The

<sup>184</sup> Of these, 29 studies have primary bio-based feedstocks that may generate iLUC in scope.

annualised area of land transformed was calculated considering a distribution period of 20 years, while the corresponding CO<sub>2</sub> emissions (accounting for changes in the content of soil organic carbon) were calculated using the method proposed by the ILCD Handbook (EC-JRC, 2010). Impacts from forest clearing were also taken into account, through an *ecoinvent* dataset specifically related to land clearing in the Amazon forest (*Provision, stubbed land, BR*). Tsiropoulos et al. (2015) estimated an upper and a lower bound for the climate impact from overall land use change emissions from both dLUC and iLUC, based on emissions factors taken from the literature (i.e. the review of - indirect- land use change models by Wicke et al., 2012). Posen et al. (2016) identified a distribution for overall land use change GHG emission factors through the already mentioned CCLUB tool (Carbon Calculator tool for Land Use Change) included in the GREET model (Wang, 2014; see Section K.2.2.2), accounting for both direct and indirect LUC. Finally, Parajuli et al. (2017) calculated GHG emissions from iLUC differently, depending on the applied LCA approach (attributorial or consequential). In the consequential assessment, the authors accounted for both the emissions due to feedstock cultivation in an already productive land in Denmark, and avoided emissions due to ultimate displacement of agriculture products from biorefinery co-products obtained from processing the feedstock grown on that land. In the attributorial assessment, no co-product substitution was applied and no avoided iLUC emissions were thus accounted (but only induced emissions from feedstock cultivation). GHG emission factors considered for iLUC directly related to feedstock cultivation were different in the two applied approaches and were in both cases taken from the literature (Schmidt and Muñoz, 2014 and Fritsche et al., 2010, respectively).

**Table K.4.** Models, guidelines or emission factors applied in the reviewed studies to quantify GHG emissions from iLUC.

<b>Model / Guideline / Emission factor (source)</b>	<b>Number of studies</b>	<b>Reference(s)</b>
Guidelines from the ILCD Handbook, accounting for soil organic carbon stock changes (EC-JRC, 2010)	1	Alvarenga et al. (2013b)
GREET model (CCLUB tool for LUC) (Wang 2009; 2014) <sup>(1)</sup>	2	Kendall (2012) Posen et al. (2016)
Default emission factors from California EPA Air Resource Board (proposed) Low Carbon Fuels Standard (California EPA, 2009 and subsequent updates)	2 <sup>(2)</sup>	Liptow and Tillman (2012) Shen et al. (2012)
Causal-descriptive modelling approach by E4tech (Bauen et al., 2010)	1	<i>Ziem et al. (2013)</i> <sup>(3)</sup>
Emission factors from literature (Righelato and Spracklen, 2007; Fargione et al., 2008; Searchinger et al., 2008; Fritsche et al., 2010; Nassar et al., 2010; Tyner et al., 2010; Piemonte and Gironi, 2011; Wicke et al., 2012; Schmidt and Muños, 2014)	9 (2 from in-depth assessment; 6 from screening assessment)	Tsiropoulos et al. (2015) Parajuli et al. (2017) <i>Gironi and Piemonte (2010)</i> <sup>(3)</sup> <i>Piemonte and Gironi (2011)</i> <sup>(3)</sup> <i>Piemonte and Gironi (2012)</i> <sup>(3)</sup> <i>Eerhart et al. (2012)</i> <sup>(3)</sup> <i>Kikuchi et al. (2013)</i> <sup>(3)</sup> <i>Suwanmanee et al. (2013)</i> <sup>(3)</sup> <i>Ganne-Chédeville and Diederichs (2015)</i> <sup>(3)</sup>
Not reported	1	<i>Leejarkpai et al. (2016)</i> <sup>(3)</sup>

<sup>(1)</sup> Applied directly in the study (i.e. in Posen et al., 2016) or relying on emission factors derived from earlier applications of the model (Kendall et al., 2012).

<sup>(2)</sup> An emission factor from the Low Carbon Fuels Standard of California was also applied in Eerhart et al. (2012), within a sensitivity analysis considering values taken also from other literature sources. The study by Eerhart and colleagues was hence classified in this table as relying on emissions factors from the literature for iLUC modelling (last row).

<sup>(3)</sup> Study not included in the in-depth assessment. Due to the limited number of selected studies addressing iLUC, this overview table has been extended also to studies that otherwise were not assessed in-depth (as only included in the screening assessment).

Beyond the studies discussed above, nine additional studies not included in the in-depth analysis addressed iLUC impacts on climate (alone or as part of overall LUC emissions), as a base case or as a sensitivity analysis (Gironi and Piemonte, 2010; Piemonte and Gironi, 2011 and 2012; Eerhart et al., 2012; Kikuchi et al., 2013; Suwanmanee et al., 2013; Ziem et al., 2013; Ganne-Chédeville and Diederichs, 2015; Leejarkpai et al., 2016). Gironi and Piemonte (2010) and Piemonte and Gironi (2011; 2012) calculated overall land use change emissions based on the results reported by Searchinger et al. (2008) and Righelato and Spracklen (2007). On this basis, a unique average emission factor of 328 metric tons of CO<sub>2</sub> eq. per hectare was considered. In Eerhart et al. (2012), the climate impact of iLUC was explored in a sensitivity analysis, applying a range of literature data (i.e. 7, 14 and 30 g CO<sub>2</sub> eq./MJ of ethanol) taken from different sources (California Low Carbon Fuels Standard; Tyner et al., 2010; Nassar et al., 2010). Kikuchi et al. (2013) quantified the iLUC GHG emission factor based on earlier results from

Fargione et al. (2008), i.e. 165 t CO<sub>2</sub>/ (50y·ha cleared). This means that the initial clearing of the land is estimated to release 165 t CO<sub>2</sub>/ha, while an amortisation period equal to 50 years is considered to calculate annual emissions, which is somehow arbitrarily chosen (e.g. IPCC suggests 20 years). In Suwanmanee et al. (2013), the average value of 328 t CO<sub>2</sub> eq. per hectare of land converted to cropland, already adopted in the abovementioned study by Piemonte and Gironi (2011), was applied as emission factor to quantify overall land use change emissions. Ziem et al. (2013) adopted the causal-descriptive modelling approach by E4tech (Bauen et al., 2010), while Ganne-Chédeville and Diederichs (2015) relied on the results from the review of land use change models by Wicke et al. (2012) to assess total land use change emissions. Finally, also Leejarkpai et al. (2016) accounted for iLUC in their study, although the approach taken for quantification of the corresponding emissions was not transparently detailed in the analysis.

#### **K.2.2.4. Modelling of waste or residual bio-based feedstocks**

Different types of waste or residual bio-based materials may be used (currently or in the future) as feedstock for (bio)-plastics production. These include, for instance, wheat straw, maize stover, wood bark and chips, sugarcane bagasse and organic matter in wastewater, just to mention a few. Therefore, the agricultural, forestry or industrial processes generating such residues as by-products, or the (biorefinery) processes receiving actual waste streams as input material<sup>185</sup>, are essentially multifunctional. The use of waste or residual feedstocks and related modelling approach was considered in a limited number of studies included in the in-depth assessment (i.e. 6 out of 32). To ensure appropriate coverage of this relevant methodological issue, the analysis was thus expanded to the whole set of studies initially collected for review, and not only limited to those selected for a more in-depth assessment. Overall, 32 studies were ultimately analysed, showing the application of different methodological approaches, while a couple of studies did not provide enough information in this respect (Akanuma et al., 2014; Kikuchi et al., 2018) (see Table K.5).

In several studies (9 out of 32), a “cut-off” approach was applied, i.e. no upstream production burdens were allocated to the feedstock, but only the full burdens associated with any subsequent collection/harvesting and recovery/processing operation carried out before actual conversion into a first useful product (e.g. in a biorefinery). This was mostly the case of studies where actual industrial waste streams or wastewater flows were used as feedstock, such as starch-rich wastewater from potato processing (Broeren et al., 2017), whey from dairy industries (Koller et al., 2013), orange peel (Günkaya and Banar, 2016), and shrimp shells (Leceta et al., 2013, 2014). However, in one screening study (Detzel et al., 2013), this approach was also applied to maize straw generated during harvesting, assigning no burdens from maize cultivation to the straw, according to the specifications provided by the Renewable Energy Directive (EC, 2009). Daful et al. (2016) and Daful and Görgens (2017) seemingly applied a cut-off approach also to sugarcane harvesting residues (leaves and tops), while in Pommeret et al. (2017) this was the case of food waste (which apparently was not assigned any upstream burden before generation). Similarly, a couple of studies focusing on wastewater treatment systems incorporating PHA production from organic matter in municipal or industrial wastewater (Gurieff and Lant, 2007; Morgan-Sagastume et al., 2016) assigned no upstream burdens to wastewater entering the system, according to the so-called “zero-burden approach”. However, these studies were aimed at comparing different process configurations for wastewater treatment, considering also the integration of PHA production or its application in place of biogas production. They hence applied a functional unit related to the provision of wastewater treatment service rather than to polymer production and supply, and a system boundary departing from waste

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<sup>185</sup> As well as processes and activities shared between consecutive life cycles, such as production of the waste feedstock and its collection and transport and processing (i.e. “recycling”) into a new useful material/product at the biorefinery. This incurs the issue of “how much” burdens from these shared processes should be allocated to the waste feedstock or to its upstream life cycle.

generation/treatment, as it is typically the case of waste-management related studies (e.g. Finnveden et al, 1999).

Economic allocation was frequently applied (in seven studies), to attribute a share of the burdens from agriculture, forestry or processing activities to wheat straw (Forte et al., 2016), forest or wood processing residues (Liptow et al, 2015; Aryapratama and Janssen, 2017), sugarcane bagasse (Daful et al., 2016; Daful and Görgens, 2017; Mandegari et al., 2017), and sawmill chips (Liptow et al., 2013). Economic allocation was also applied to wheat straw by Parajuli et al. (2017), when adopting an attributional LCA approach. Conversely, mass allocation was applied only in three studies to maize stover (Yu and Chen, 2008), forest residues (Chen et al., 2016), and pulp mill side streams (Gontia and Janssen, 2016). Allocation based on mass (and on physical criteria in general) was considered by some authors inappropriate for agricultural and forestry residues like straw and bark (Ahlgren et al., 2015), as these are usually produced in a fixed proportion with the main product (grains or debarked wood) and thus a mass-based allocation would not reflect any underlying physical relationship among co-products.

When agricultural residues like maize stover and wheat straw were used as feedstock, system expansion was also applied (4 studies), accounting for the consequences of redirecting the feedstock from its current fate to the biorefinery, in line with a more consequential approach (Kim and Dale 2005; Tao et al., 2014; Dunn et al., 2015; Adom and Dunn, 2017). According to this approach, only the burdens from collecting the residue from the field and subsequent baling and transport to the biorefinery were accounted, along with the potential consequences of removing the feedstock from the field. In this respect, all the mentioned studies except for Kim and Dale (2005) assumed the replacement of removed residues with synthetic fertilisers for nutrient supply, while Kim and Dale (2005) accounted for changes in soil organic carbon (SOC) due to straw removal. However the latter did not specify how the residual feedstock was actually handled. System expansion was also applied to wheat straw by Parajuli et al. (2017), when implementing a consequential LCA approach. In this case, beyond straw replacement with mineral fertilisers and changes in SOC levels considered in the studies above, the assessment also accounted for net (nitrogen) emissions resulting from straw replacement. Finally, Kendall (2012) applied system expansion to cellulosic residues from material recovery facility (MRF) used as a feedstock for PHB production. As in the previous studies dealing with agricultural residues, no upstream burdens were assigned to the MRF residues, but the consequences of diverting them from their current alternative fate (in this case landfilling was assumed) were taken into account and factored in the model as avoided burdens from landfilling.

In Kurdikar et al. (2000), system expansion was applied via substitution to the case of genetically modified maize stover used as a feedstock for PHA production. In this case, the main product from harvesting (i.e. genetically modified maize grain), was assumed to replace an equivalent quantity of conventional grain from non-genetically modified production. However, mass-based allocation was also applied as a sensitivity analysis.

Beyond the already mentioned study by Parajuli et al. (2017; see above), also Heimersson et al. (2014) applied or discussed different approaches to deal with multi-functionality associated with the use a waste/residual feedstock, focusing on the case of pure- or mixed-culture PHA production from organic waste flows or organic matter in wastewater. Explored approaches were cut-off, a zero-burden approach, allocation based on appropriate criteria, and system expansion (to account for the consequences of redirecting the waste feedstock to PHA production). The choice was affected by the specific situation applying to the feedstock (e.g. being a real waste or a by-product with positive economic value) and by the LCA approach adopted in the analysis (attributional or consequential).

Finally, a few studies applied as such the default approach implemented in the used secondary datasets (Guo and Crittenden, 2011; Rostkowski et al., 2012; Karka et al.,

2017). These were in most cases based on cut-off (i.e. for corn stover and biogas from sewage sludge treatment), or economic allocation (wheat straw and residual wood used for woodchips production).

**Table K.31.** Approaches and methods applied in the reviewed studies to handle multi-functionality associated with the use of waste or residual feedstocks for bio-based plastics production. (References reported in *italic* were not included in the in-depth assessment)<sup>186</sup>.

Approach	Number of studies	Reference(s)
Cut-off	9	<i>Detzel et al. (2013)</i> <i>Koller et al. (2013)</i> <i>Leceta et al. (2013)</i> <i>Leceta et al. (2014)</i> Daful et al. (2016) <sup>(1)</sup> <i>Günkaya and Banar (2016)</i> <i>Broeren et al. (2017)</i> <i>Daful and Görgens (2017) (1)</i> <i>Pommeret et al. (2017) (2)</i>
Zero-burden approach	2	<i>Gurieff and Lant (2007)</i> <i>Morgan-Sagastume et al. (2016)</i>
System expansion <sup>(3)</sup>	5	<i>Kim and Dale (2005) (4)</i> Kendall (2012) <i>Tao et al. (2014)</i> <i>Dunn et al. (2015)</i> <i>Adom and Dunn (2017)</i>
System expansion (via substitution)	1	<i>Kurdikar et al. (2000) (5)</i>
Mass-based allocation	3	<i>Yu and Chen (2008)</i> Chen et al. (2016) <i>Gontia and Janssen (2016)</i>
Economic allocation	7	Liptow et at. (2013) <i>Liptow et al. (2015)</i> Daful et al. (2016) <sup>(6)</sup> Forte et al. (2016) <i>Aryapratama and Janssen (2017)</i> <i>Daful and Görgens (2017) (6)</i> <i>Mandegari et al. (2017)</i>

<sup>186</sup> Due to the limited number of selected studies addressing the use of waste/residual bio-based feedstocks and related modelling approaches, this overview table has been extended also to studies that otherwise were not assessed in-depth (as only included in the screening assessment).

Approach	Number of studies	Reference(s)
Different methods	2	<i>Heimersson et al. (2014)</i> <sup>(7)</sup> <i>Parajuli et al. (2017)</i> <sup>(8)</sup>
Default approach implemented in the applied dataset (cut-off or economic allocation)	3	<i>Guo and Crittenden (2011)</i> <sup>(9)</sup> <i>Rostkowski et al. (2012)</i> <sup>(10)</sup> <i>Karka et al. (2017)</i> <sup>(11)</sup>
Not specified	2	<i>Akanuma et al. (2014)</i> <i>Kikuchi et al. (2018)</i>

<sup>(1)</sup> Cut-off was applied to sugarcane residues (leaves and tops), while economic allocation was applied to sugarcane bagasse produced during sugarcane processing (as reported below in the table).

<sup>(2)</sup> The applied approach was not clearly specified in the study, but it was inferred that food waste entering the biorefinery was attributed no upstream environmental burdens occurring before its generation, following a cut-off approach.

<sup>(3)</sup> Here it is intended as the modelling of the consequences of re-directing the residual agricultural feedstock (wheat straw or maize stover) from its current fate to the biorefinery, along with the burdens from operations performed to make it available to the biorefinery itself (e.g. removal from the field and transport).

<sup>(4)</sup> The applied approach was not specified in the study, but it was inferred from the text considering that only the burdens from stover harvesting and transport were accounted for, along with potential effects on soil organic carbon due to stover removal from the field.

<sup>(5)</sup> Mass-based allocation was also applied as a sensitivity analysis.

<sup>(6)</sup> Economic allocation was applied to sugarcane bagasse produced during sugarcane processing, while sugarcane residues (leaves and tops) were handled via cut-off (as reported above in the table).

<sup>(7)</sup> The study explored or discussed different approaches to deal with the use of organic waste flows or organic matter in wastewater as a feedstock for pure- or mixed-culture PHA production. These approaches were affected by the specific situation applying to the waste feedstock and by the chosen LCA approach (attributorial or consequential), and included a cut-off/zero-burden approach, allocation based on appropriate criteria and system expansion (to account for the consequences of redirecting the waste feedstock to PHA production).

<sup>(8)</sup> Depending on the applied LCA approach (attributorial or consequential). In attributorial LCA, economic allocation was applied between wheat straw and grains co-produced during harvest, with straw being used as a feedstock for the product. In consequential LCA, system expansion was applied, accounting for the effects of removing straw from the field and diverting it to the biorefinery (i.e. replacement of nutrient supplied with straw by application of synthetic fertilisers, net N emissions resulting from this replacement, and emissions due to changes in soil organic carbon). Removal, bailing and transport operations were also taken into account.

<sup>(9)</sup> Applying cut-off to corn stover, which is assigned no upstream burdens from cultivation. Stover harvesting was included in the study.

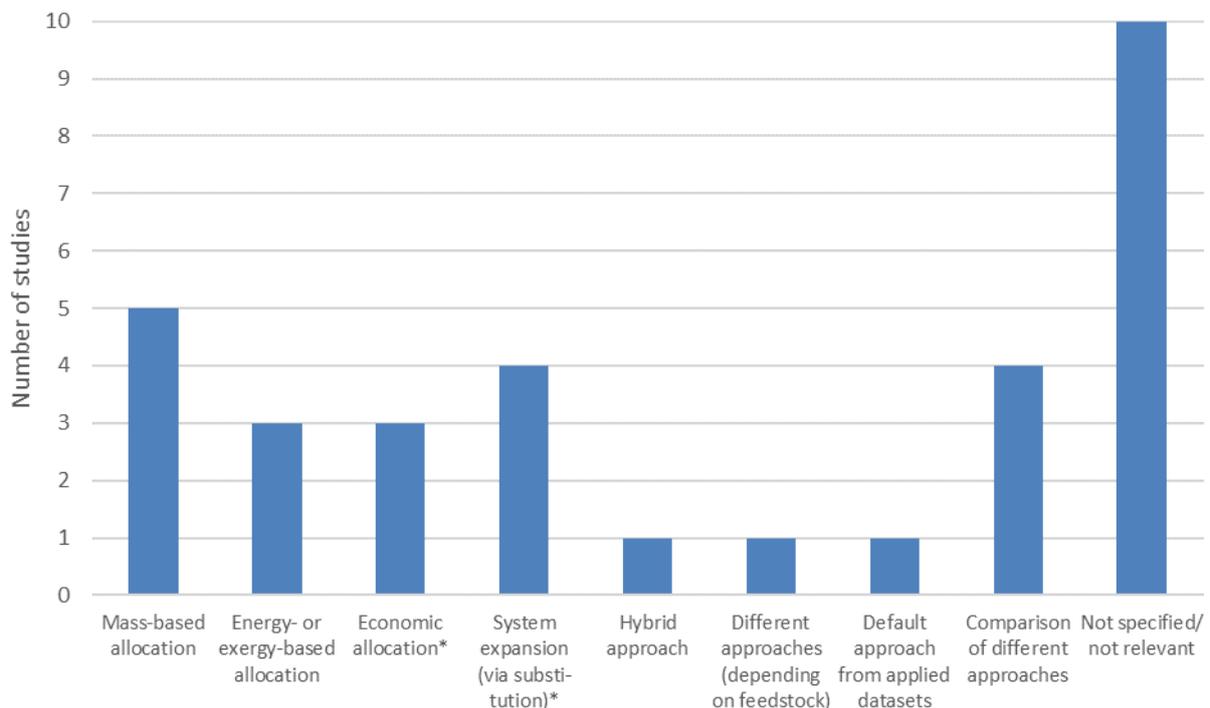
<sup>(10)</sup> Applying cut-off to biogas from sewage sludge, which is assigned no upstream burdens from wastewater treatment or further upstream.

<sup>(11)</sup> Applying economic allocation to wheat straw (and grains), and to residual wood used for woodchips production.

#### **K.2.2.5. Handling of multi-functionality at multi-output biorefineries**

Regarding the handling of multi-functionality at the biorefinery level due to production of multiple outputs, it was observed that 21 out of the 32 studies included in the in-depth analysis specified the approach(es) used to address this methodological aspect (Figure K.4). Conversely, 11 studies either did not specify/give further details on this issue, simply applied as such available datasets for biopolymers (Parker and Edwards, 2012)), or did not directly include (multifunctional) biorefinery processes in the analysed system(s) (i.e. Forte et al., 2016; Shen et al., 2012 and Gérard and Roux, 2014, whit the latter not addressing any bio-based products). Most studies (15) applied only one approach, or a combination of these (hybrid approach), for a given process, although one of these studies applied a different approach depending on the feedstock used in the biorefinery (sugarcane or willow; Van Uytvanck et al., 2014). Conversely, in 6 studies

alternative approaches were evaluated for a same process, as sensitivity analysis (2 studies; Kim and Dale, 2008 and Razza et al., 2015; not shown separately in Figure K.4 but only considering the approach applied as a base case), or as part of a comparison among different LCA approaches or scenarios (4 studies; Liptow and Tillman, 2012; Tsiropoulos et al., 2015; Posen et al., 2016 and Parajuli et al., 2017). Excluding sensitivity analyses and studies comparing different approaches/scenarios, mass-based allocation was the most frequently applied approach (5 studies; Detzel and Krüger, 2006; Deng et al., 2013; Van der Harst et al., 2014; Razza et al., 2015; and Hottle et al., 2017). System expansion through substitution was applied in a similar number of studies (i.e. four)<sup>187</sup>, although these also become five when considering the system expansion approach adopted for one of the two biorefinery configurations addressed in Van Uytvanck et al. (2014; production of willow-based ethanol). Economic allocation was applied in three studies<sup>188</sup> (four if considering sugarcane processing in Van Uytvanck et al., 2014), while other three studies applied allocation based on energy content (Belboom and Léonard, 2016; Markwardt et al., 2017), or exergy (Alvarenga et al., 2013a). Groot & Borén (2010) applied a hybrid approach combining different methods for the different co-products of a sugar mill (i.e. economic allocation based on selling price for valuable physical by-products, and substitution for electricity generated from bagasse combustion). Regarding those studies comparing different LCA approaches, two of them applied both economic allocation (attributional LCA) and system expansion (consequential LCA) (Tsiropoulos et al., 2015 and Parajuli et al., 2017), while Liptow and Tillman (2012) compared results calculated applying either system expansion or energy-based allocation. Finally, Posen et al. (2016) evaluated alternative scenarios applying different viable approaches, including mass-based allocation, energy-based allocation and system expansion.



(\*) Economic allocation and system expansion via substitution were also individually applied in the study accounted in column #6, and adopting different approaches depending of the feedstock processed in the biorefinery.

**Figure K.4.** Approaches and methods applied in the reviewed studies to handle multi-functionality at the biorefinery level due to production of multiple outputs.

<sup>187</sup> Including Kim and Dale (2008); Alvarenga et al. (2013b); Liptow et al. (2013); and Chen et al. (2016).

<sup>188</sup> Including Guo et al. (2013); Hansen et al. (2015); and Daful et al. (2016).

#### **K.2.2.6. Additives: consideration, modelling and impacts**

Several additives are normally used in plastic production to obtain a material or product with suitable (minimum) technical performances. They include, for instance, mineral fillers, plasticisers, flame retardants, impact modifiers, reinforcing agents, heat stabilisers and colorants (OECD, 2009; Broeren et al, 2016; ECHA, 2019). Additives are generally mixed with melted polymeric material during product manufacturing (e.g. extrusion or blow moulding), although they may also be added upstream during the production of polymer pellets/granulate (e.g. anti-oxidants) or when more polymers are compounded together (e.g. compatibilisers used in polymer blends such as starch-based or PLA-based polymers).

Additives were explicitly considered only in the studies by Broeren et al. (2017 and 2016), both excluded from the in-depth assessment, but reported and investigated here to cover the issue of whether and how additives were addressed so far in the reviewed literature. According to Broeren et al. (2016), additives account for about 10% of the weight of plastics on average<sup>189</sup>, thus being potentially relevant for consideration in environmental assessment and possibly having a non-negligible influence on the respective results. However, the actual quantities and specific substances used as additives for plastics are rarely disclosed, representing an uncertainty that is typically ignored in LCA studies of final products (Broeren et al., 2016). For starch-based plastics, additives such as compatibilisers, plasticisers, processing aids and fillers are typically applied, to achieve a favourable balance between technical properties, processability, and cost. They normally account for about one third (25-30%) of the final weight of starch plastics, and can thus strongly affect their overall environmental performance (Broeren et al., 2017; IfBB, 2018).

In the screening LCA study by Broeren et al. (2016), four families of additives were distinguished for each of the nine plastic grades assessed: mineral fillers, plasticisers, flame retardants and others/unknown. The more general focus on additive categories, rather than on the specific substances used, avoided the collection or determination of full composition data, which are normally hardly retrievable, at least when investigating new material alternatives at the screening level. Depending on the considered plastic grade, additives accounted for different ranges (from 0-10% to 31-40% by weight), although quantities were not explicitly reported (nor the specific share of single additives categories). To overcome the absence of detailed LCI data for many substances, representative ranges of non-renewable energy use and GHG emissions associated with the production of each family of additives were derived from the literature, and applied in the assessment to calculate overall impact ranges for each plastic type (grade). Applied values referred to specific representative substances of each additive category, or to a combination of these substances. Burdens from additive use occurring in other lifecycle stages than production (e.g. from release during the Use stage or End of Life) were not considered. For plastic grades including flame retardants, additive production was found to account for 5-25% to 10-45% of cradle-to-grave GHG emissions, depending on the plastic grade and the value considered for the GHG emission factor of additive production within the assumed range. For non-flame retardants grades, the contribution was lower (up to 5%), instead.

The six grades of starch-polyester plastic studied in Broeren et al. (2017) included 16% to 32% of additives by weight, distinguished between compatibilisers and other additives (mineral fillers, plasticisers, processing aids). Their production was modelled by combining confidential data on substances and related production burdens with LCI datasets from commercial databases, and no further detail was provided. Additive production was shown to account for up to 46% of total cradle-to-gate non-renewable energy use and GHG emissions for those plastic grades using larger amounts of additives in the range of 30% (e.g. starch/PLA). When smaller amounts of additives were used in

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<sup>189</sup> Based on overall additive use and total plastic production in 2005, which amounted to about 22 and 225 million tonnes, respectively (Van Oers et al., 2012; PlasticsEurope, 2013).

combination with relatively energy-intensive polyesters (e.g. as in starch/PBS), additive production only accounted for 9% of those indicators. Production of additives also provided a non-negligible contribution to eutrophication and agricultural land use, when they were of bio-based origin.

#### *K.2.2.6.1 Use of additives to improve a specific product (packaging) performance*

A separate issue is represented by the use of additives in (finished) plastic products (usually packaging) in the attempt to further improve their performance with respect to a specific functionality (e.g. food preservation and shelf life extension). In this case, additives are generally not mixed directly with the plastic material during polymer production, compounding or product manufacturing to ensure suitable (minimum) technical properties to the final material or product (i.e. they do not necessarily need to be used to obtain a suitable material/product for the intended application). Conversely, they are generally applied to the finished product once already formed (although they may also be added to melted plastic material during product manufacturing) in the attempt to provide additional functionalities to the product (i.e. their use is intentional but not necessary). While this possibility is not the primary focus of this section, a brief discussion is reported below, with reference to exemplary situations reported in some of the studies collected for the review (and not included in the in-depth assessment).

In the study of Lorite et al. (2017), a number of PLA-based nanocomposite active packaging alternatives were prepared, by means of different additives, and their environmental performance was compared with that of PET and pristine PLA packaging. Studied PLA-based active packaging alternatives were composed of (i) PLA and nanoclay; (ii) PLA, nanoclay and surfactant; and (iii) PLA and nano-whiskers. Inventory data related to production of additives were derived from laboratory tests. All additives were assumed to extend the lifetime of the packaged product (fruit) equally long. The results showed that PLA-based packaging with nano-whiskers additives requires more energy and materials, and presents the largest impact in all impact categories, compared to the other investigated PLA-based solutions. However, the additives had only a minor influence on the material demand compared to pristine PLA and PET pellets. Although PLA-based packaging alternatives including additives required more energy for their production, results clearly showed that PLA-based packaging with nanoclay and surfactant is very competitive with respect to PET packaging in terms of environmental impact. Moreover, PLA-based packaging alternatives generated a lower potential impact on human health compared to PET packaging. Finally, the results indicated that PLA-based packaging with additives can reduce food loss through extending the shelf-life of the product (fruit).

Petrucci et al. (2017) assessed the potential environmental impact of a limonene (20% wt.) plasticized PLA film containing 1% wt. of cellulose nanocrystals (PLA/1% CNC/20% limonene) and compared it to the impact of a PLA film containing 3% wt. of organo-modified montmorillonite (OMMT) plasticized with acetyl tributyl citrate (ATBC) at 16.5% wt. Data related to the extraction of cellulose nanocrystals were determined at laboratory scale, while data regarding the production of limonene, used as a plasticizer in the PLA/CNC-based film, were collected from a specific process carried out in a 200,000 ton/yr biorefinery. According to the results, even if the cellulose nanocrystals amount used in the film was relatively low, its contribution to total environmental impacts was relevant. This was mainly due to the use of a wide range of chemical substances (i.e. sodium hydroxide, sodium dichlorite, sodium bisulphate, acetic acid, etc.) required for the extraction of cellulose nanocrystals from Phormium fibres. The contribution of limonene to the environmental impacts was generally not significant, with a maximum of 13% in the case of climate impact. A slightly better environmental performance was associated with the limonene plasticized nanocomposite PLA film based on cellulose nanocrystals, with respect to ATBC plasticized film containing organo-modified montmorillonite.

#### **K.2.2.7. Modelling of biodegradation at End of Life (biological treatment or in-situ)**

Twelve studies included in the in-depth analysis addressed product biodegradation at the End of Life stage. This included either (controlled) biodegradation during organic waste treatment processes such as aerobic composting and anaerobic digestion (10 studies), or in-situ degradation (i.e. on or into the soil; 2 studies). While (partial) biodegradation may occur also during landfilling, it was excluded from the scope of this section, as landfill deposition is not a treatment specifically intended to achieve biodegradation, but rather a disposal option.

All the studies dealing with biodegradation applied product-specific (or better, material-specific) biodegradation rates<sup>190</sup>, rather than average degradation efficiencies typically achieved for generic organic waste. In most cases (5 studies), literature values from previous LCA or experimental studies were considered (Guo et al., 2013; van der Hast et al., 2014; Rossi et al., 2015; Posen et al., 2016; Hottle et al., 2017). Assumptions of complete or partial biodegradation were also frequently performed (4 studies, including Detzel and Krüger, 2006; Arnold and Alston, 2012; Kendall, 2012; and Deng et al., 2013). Only a couple of studies explicitly relied, at least partially, on results from laboratory tests prescribed in relevant compostability standards such as EN 13432 (Müller, 2012; Razza et al., 2015), while in one case study on mulch film (Müller and Müller, 2015), the minimum biodegradation rate required by the European standard for biodegradable plastic mulching (at the time under development) was assumed (i.e. 90% over 2 years; EN 17033).

Due to the diversity of sources and assumptions considered in the studies to determine biodegradation rates, the values applied to a given material for a specific end-of-life option frequently varied over a quite wide range, especially in the case of industrial composting (Table K.6). For instance, biodegradation of PLA in industrial composting varied between 44% (Posen et al., 2016) and 95% (Detzel and Krüger, 2006), similarly to that of PHB (44-100%). For starch blends (starch/copolymers) and thermoplastic starch (TPS), the range was more restricted, albeit still appreciable (60-80% for TPS and 75-91% for starch blends). A similar situation was observed for anaerobic digestion, with a more marked variation for PLA (60-98%) and less pronounced for starch blends (68-75%). Regarding in-situ (bio)-degradation, Arnold and Alston (2012) applied interesting assumptions for conventional PP and PLA/starch biopolymer used in tree shelters, as detailed in Table K.6. In particular, a 50% initial degradation of PP shelters to CO<sub>2</sub> and water was assumed, based on empirical observations on shelters disintegration (reducing their mass by nearly one half at the point in which they lose mechanical integrity) and literature data for polymer coatings. However, this assumption has to be taken carefully, as mass losses from mechanical disintegration may not be necessarily associated to an actual degradation to CO<sub>2</sub> and water as observed for (much) thinner polymer coatings.

Among the studies not included in the in-depth analysis, it is worth mentioning the one by Hermann et al. (2011), where average biodegradation rates for a number of biopolymers under industrial composting conditions were determined based on an extensive survey of the literature available at the time of the study. Surveyed biopolymers included TPS, PLA, a starch blend (MaterBi), PBAT, PHB and PHBV.

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<sup>190</sup> Generally intended as (carbon) mineralisation rates, i.e. as the percentage of carbon in the product or material that is ultimately converted to CO<sub>2</sub> and or CH<sub>4</sub> during biodegradation.

**Table K.6.** Biodegradation or degradation rates applied in the reviewed studies for different plastic materials and End of Life options. Values refer only to (bio)-degradation during the relevant specified option.

Polymer	End of Life option				Source
	Industrial composting	Home composting	Anaerobic digestion	In-situ (bio)-degradation	
PLA	44-95%	-	60-98%	-	Detzel & Krüger (2006); Rossi et al. (2015); Posen et al. (2016); Hottle et al. (2017); Van der Harst et al. (2014)
PLA/copolymer <sup>(1)</sup>	100%	-	-	90% <sup>(2)</sup>	Müller (2012); Müller & Müller (2015)
TPS	60-80%	-	90%	-	Rossi et al. (2015); Hottle et al. (2017)
Starch/copolymers	75-91%	86-90%	68-75%	-	Guo et al. (2013); Razza et al. (2015)
Starch/PLA	-	-	-	50% <sup>(3)</sup>	Arnold & Alston (2012)
PHB	44-100%	-	-	-	Kendall (2012); Posen et al. (2016)
Wheat gluten	100%	-	-	-	Deng et al. (2013)
PP	-	-	-	50% <sup>(4)</sup>	Arnold & Alston (2012)

<sup>(1)</sup> Refer to a blend of PLA and PBAT (also traded under the commercial name of *ecovio*<sup>®</sup>).

<sup>(2)</sup> The remaining 10% was assumed to be bound in the non-toxic form in clay as soil organic matter.

<sup>(3)</sup> The remaining 50% was assumed to be degraded to sugars without any additional environmental impact. Pigments and stabilisers do not degrade and were considered as emitted to soil.

<sup>(4)</sup> 50% was assumed to be lost as CO<sub>2</sub> and water, the remaining 50% was degraded by photo-oxidation to aldehydes and ketones (50%) organic acids (20%), esters (20%) and alcohols (10%). Pigments and stabilisers do not degrade and were considered as emitted to soil.

Regarding the modelling of biodegradation within specific processes (i.e. composting and anaerobic digestion), the focus of the investigated studies was mostly on the fate of carbon and (any) nitrogen included in the biodegradable plastic during treatment and subsequent on-land application of the residual composted or digested material (if performed). Hence, biodegradation was mostly addressed in terms of carbon (CO<sub>2</sub> and CH<sub>4</sub>) and nitrogen (NH<sub>4</sub> and N<sub>2</sub>O) emissions taking place during these activities. The modelling also generally accounted for any avoided processes or emissions (e.g. from avoided mineral fertilisers application) associated with the carbon and nitrogen possibly transferred to the residual composted or digested material, and remaining available in soil after on-land application and further degradation. The details of the modelling of biodegradation after on-land application were frequently not reported across the set of reviewed studies and, in general, no common and consistent approach could be identified in this respect. However, it can be reasonably inferred that the fate and release of any non-biodegradable substances or compounds originally present in the biodegradable plastics (e.g. metals and additives) were usually not addressed, nor the possible generation and release of any intermediate degradation product (including micro-plastics). Information about full product composition and use of additives are

indeed scarcely available for accounting in LCA studies, and bio-degradation pathways still have to be completely understood.

As for the overall modelling of organic waste treatments where biodegradation occurs, and especially aerobic composting, the majority of the studies investigated in the in-depth assessment applied an approach that can be assimilated to the "individual perspective" described by Hermann et al. (2011). According to this approach, the biodegradable plastic is considered in isolation from any other co-processed organic waste (e.g. bio-waste from municipal collection), and only the burdens and possible benefits associated with its treatment are taken into account. This means that only the emissions directly related to the degradation of the biodegradable plastic are modelled, based on its specific composition and degradation rate (e.g. no nitrogen emissions are considered if nitrogen is not present in the original composition of the product or material sent to composting). Similarly, any benefits from compost utilisation are calculated considering the actual estimated quantity and composition of residual organic material that can be realistically obtained from biodegradable plastic composting (i.e. no benefits from replacing mineral fertilisers are modelled if the biodegradable plastic does not contain any nutrients that can be transferred to the compost and ultimately applied on land). Finally, process-specific burdens associated with waste handling and treatment operations (e.g. from electricity and fuel consumption) are also taken into account, generally in proportion to the mass of biodegradable plastic waste to be treated (according to common practice applied to composting modelling).

Conversely, in Razza et al. (2015), composting of biodegradable plastic was modelled according to a variant of the "combined perspective" discussed by Herman et al. (2011). Following this approach, a product-specific biodegradation rate (75%) was considered, and the corresponding carbon emissions were modelled, as in the "individual perspective" described above. However, a share of nitrogen emissions ( $\text{NH}_3$ ,  $\text{N}_2\text{O}$ ) normally occurring during organic waste composting were assigned to the biodegradable plastic, based on average emission data reported for such process. Moreover, benefits from compost utilisation were taken into account, and calculated considering that the residual organic material obtained from the bio-plastic has the average characteristics (e.g. nutrient content) of compost commonly obtained from organic waste treatment (thus allowing for the replacement of mineral fertilisers, other than for peat substitution and carbon sequestration). Similarly, Detzel and Krüger (2006) mentioned that compost from PLA degradation was assumed to serve as soil amendment and to displace mineral fertilisers and peat. However, no sufficient information was provided in the study to understand how the overall modelling of such displacement and of the composting process was conducted. On the other hand, Müller (2012) did not account for any benefits from potential fertiliser substitution after composting of biodegradable plastic carrier bags, as these were assumed to entirely decompose to  $\text{CO}_2$  and water.

#### ***K.2.2.8. Potential consequences and indirect effects of using biodegradable (compostable) plastic bags and tableware on organic waste management***

A few studies addressed the potential consequences of using biodegradable plastic products (i.e. compostable bags and cutlery) on the management of other, somehow very closely connected waste streams (i.e. food/organic waste), following a consequential-oriented approach. For carrier/grocery bags, this was the case of two studies carried out on behalf of BASF (Müller, 2012; Müller and Müller, 2017), both included in the in-depth assessment. Conversely, Razza et al. (2009) and Fieschi and Pretato (2017) addressed the same issue for compostable cutlery and tableware, respectively. Although these studies were not included in the in-depth assessment, they are equally reported below, to cover more broadly the modelling aspect discussed in this section.

In Müller (2012), the potential effects of an expected increase in separate collection of household organic waste following the use of biodegradable (i.e. compostable) carrier bags were accounted for in the LCA of such type of bags (considering their multiple use

for transport of staple goods and subsequent collection of organic waste). A 2% increase was assumed compared to the use of conventional, non-biodegradable PE bags<sup>191</sup> and paper bags. This was based on the results of a 3-months pilot project involving around 21,000 households from two districts in Berlin, which in September 2011 shifted from non-biodegradable to biodegradable bags. At the end of the three months period, a 10% increase in the amount of organic waste collected was observed compared to the project start, while a 30% increase was measured with respect to other districts not relying on biodegradable bags. An average 20% increase per user was thus assumed, along with an overall usage rate of biodegradable bags equal to 10% of total German population (corresponding to an overall 2% increase nationwide). The effects of this limited increase on the overall LCA (eco-efficiency) results was only marginal, though, and differences among the analysed alternatives were small and not significant. The study also claimed potential environmental benefits from a possible reduction in the level of non-biodegradable plastic impurities (from conventional PE bags) present in the collected organic waste and entering the subsequent composting treatment, according to the results of the pilot project mentioned above<sup>192</sup> and the discussion reported by Razza and degli Innocenti (2012). However, this effect was not taken into account in the study.

Also Müller and Müller (2017), who focused more specifically on fruit and vegetable bags, accounted for an increase in the organic waste capture rate when compostable bags are used (+36% compared to conventional non-biodegradable bags and +14.5% compared to paper bags). Moreover, the analysed type of biodegradable bag was reported to potentially reduce fruit and vegetable waste generation from households compared to conventional PE and paper bags, thanks to increased shelf life of stored food. This was taken into account in three sensitivity scenarios considering alternative types of vegetables (tomato and lettuce) and different consumption patterns (linear consumption over different timeframes). The modelling was based on the results of shelf-life tests carried out for BASF, which showed a four-fold increase in tomato shelf life compared to using (closed) PE bags, and a seven-fold increase for lettuce. The study also mentioned that a higher biogas production potential can be expected when biodegradable bags are used and collected waste is sent to anaerobic digestion (i.e. up to 50% compared to collection without compostable bags), because of better conservation of discarded food waste, which prevent methanisation from starting already at households. However, it is not clear whether this effect had eventually been taken into account in the assessment or not.

Finally, Razza et al. (2009) and Fieschi and Pretato (2017) accounted for the potential effects of using compostable plastic cutlery or tableware on the collection and management of waste from fast food restaurants and canteens or from events and quick service restaurants and catering. When compostable items are used, a single waste stream including discarded cutlery/tableware and food residues can be separately collected for composting, rather than being routed to incineration or disposal as mixed residual waste when non compostable articles are used (and these are not separately collected from food residues). To account for this potentially beneficial effect, the management of all the waste generated after meal supply and consumption was included in the functional unit of the study, considering both discarded cutlery/tableware and the associated amount of food residues (assumed to be 0.15 kg per meal). A different End of Life scenario was then considered, depending on the type of cutlery or tableware used (i.e. composting for biodegradable cutlery/tableware and a mix of incineration and landfilling for non-biodegradable ones).

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<sup>191</sup> Note that non-biodegradable PE bags, after use for transport of staple goods, were assumed to be used for organic waste collection only at household level, while subsequent delivery of collected waste to the municipal collection system was made through dedicated bins without the use of bags (which are separately sent to incineration). The cleaning of bins is included in the assessment.

<sup>192</sup> A reduction of PE impurities ranging from 37% to 67% was observed, depending on the district.

#### **K.2.2.9. Other potential consequences/indirect effects associated with the use of biodegradable plastics**

Other potential consequences associated with the use of biodegradable plastics were explored for agricultural mulch film in a case study carried out for BASF (Müller, 2015). The study compared the use of conventional, non-biodegradable PE film and biodegradable *ecovio* mulch film for cotton growing in China (Xinjiang region), based on a functional unit of 1000 kg of cotton grown in the region (and hence not focusing directly on the function provided by mulch film as a defined plastic product). Both types of film were assumed to be used in the same thickness (10 µm) considering common practice adopted at the time of the study in China, and forthcoming regulation in the Country, establishing a specific thickness for mulch film. Such a limited thickness was associated with a relatively low collection rate from the field after use (i.e. 33%) for conventional non-biodegradable film, generating problems of accumulation in soil and resulting quality degradation over the years, with an ultimate expected detrimental effect on crop yield. The potential effects of both biodegradable and conventional film on cotton yield were thus taken into account in the study, considering an overall period of 15 years. In the case of non-biodegradable mulch film, an average yield decrease by 8.5% over 15 years was considered, assuming a linear correlation between measured quantities of mulch film remaining in the soil and observed yield decrease (based on CAAS, 2013 –reference not provided in the study). For biodegradable film, no yield decrease was assumed, as the film was not expected to accumulate in soil with detrimental effects on its quality. The assumed yield decrease was identified as a main driver for the results, which showed a better overall performance of cotton grown applying biodegradable film. To increase their robustness, the study clearly recommended to validate the data related to yield decrease in further field tests, and to define a solid calculation method to quantify the correlation between non-biodegradable film remaining in soil and the corresponding yield decrease.

#### **K.2.2.10. Littering and micro-plastics generation and impacts**

Littering generation and impacts were addressed (semi-quantitatively) only in one study included in the in-depth analysis, i.e. Parker and Edwards (2012), who compared the use of conventional HDPE bags, oxo-biodegradable HDPE bags and biodegradable (starch-polyester) bags in the UK. The impact of plastic bag littering were assessed at two different levels. First of all, the impacts from degradation of littered bags in the open terrestrial environment were assessed, using LCI data related to disposal of bag material(s) in a sanitary landfill as a proxy. Default data were adjusted to account for carbon emissions from degradation under aerobic environmental conditions (instead of semi-anaerobic landfilling conditions), assuming specific degradation rates. These were equal to 91% for oxo-degradable bags and 50% for biodegradable bags, while no degradation was assumed for conventional HDPE bags. In addition, an attempt was made to quantify the visual/aesthetic impact of littering, by introducing an additional indicator termed "*litter - aesthetics*", which was initially developed by ExcelPlas Australia et al. (2004). The indicator is calculated as the area of the littered product multiplied by the time spent in the environment before degrading (m<sup>2</sup>·y), both considered relevant variables in relation to aesthetic impacts. The quantification of all the mentioned impacts requires a prior estimation of the quantity of product littered at End of Life (per functional unit), by means of a specific littering rate. This was estimated by combining data from a beach survey (quantifying the relative abundance of single product categories among littered items found on the beach), with the overall littering generation in the reference country (UK), and the total amount of product (bags) consumed during the same year of the survey in the same country. An estimated share of littered bags (littering rate) equal to 0.75% was ultimately calculated based on this procedure.

In the original study by ExcelPlas Australia et al. (2004), a second indicator termed "*litter - marine biodiversity*" was proposed, in the attempt to quantify the potential impact of littering on marine organisms, but it was not taken into account in Parker and Edwards (2012). As reported in Dilli (2007), a study excluded from the in-depth

assessment, the indicator is based on the time over which the littered product has the potential for ingestion or entanglement of marine fauna (kg·y), and is mostly affected by the propensity of the material to float or sink. In Dilli (2007), 0.5% of post-consumer bags were assumed to enter the litter stream at End of Life, based on existing data related to single-use HDPE bags. However, only a qualitative assessment of the indicator was performed (via qualitative descriptions of magnitude), and no sufficient detail was provided to make this approach transparent or even reproducible.

In Müller (2012), littering was addressed through qualitative statements, highlighting that BASF does not see the use of biodegradable bags in itself as a solution to marine littering, which can only be solved by means of education.

Micro-plastics generation and impacts were not addressed in any of the studies reviewed in the in-depth analysis, nor in those considered in the screening assessment.

### **K.3. Conclusions**

Through an extended literature search, 171 studies concerning LCA applied to plastics were collected, under the condition that at least one alternative feedstock or material for plastic production was addressed. An initial classification and screening of all collected studies was then performed, by recording information on a number of general aspects related to goal and scope, level of detail, compliance with existing international LCA standards, and review according to the latter. After this screening, 32 studies were specifically selected, based on different criteria, for an in-depth analysis of methodological and modelling choices applied for a number of relevant aspects related to LCA of plastic products. The selection, apart from the overall relevance of the study for the purposes of the project (in terms of goal and scope), was based on the following criteria: inclusion of at least two alternative products or scenarios, assessment of at least one midpoint environmental impact category, availability of characterised midpoint LCIA results (expressed in physical units), as well as availability of a background report or supplementary information adequately detailing the modelling approach and the LCI data used (or at least of an extensive description of such methods and data).

Overall, the review revealed that a significant variety of approaches, assumptions and/or data was applied to handle the methodological and modelling aspects evaluated in the analysis, confirming the real need to define common and harmonised rules for increased consistency and reproducibility of LCA studies on plastics. At the same time, (very) scarce consideration of some specific aspects was detected, highlighting the importance of improving the related knowledge base and of developing suitable and sufficiently robust approaches and methods to address those issues. The main findings for the most relevant methodological and modelling aspects considered in the in-depth analysis are summarised below.

- Biogenic carbon emissions and removals from the product were addressed to different extents and levels of detail across the reviewed studies dealing with at least one bio-based feedstock (i.e. all studies included in the in-depth analysis except one). An explicit biogenic carbon balance/inventory was rarely reported (three studies), while biogenic carbon emissions and removals were modelled/accounted differently across the inventories of the reviewed studies, reflecting the absence of a common approach to deal with biogenic carbon content in products. In this respect, cradle-to-gate studies were almost evenly split in two categories, with one of them not quantifying biogenic carbon emissions and removals, and the other one accounting for the removal of biogenic carbon (CO<sub>2</sub>) incorporated in the product (as an uptake during biomass cultivation or, less frequently, as a negative emission credited to the product system). A similar situation was also observed for full cradle-to-grave studies and cradle-to-gate studies including also End of Life, with nearly two thirds of these accounting for the removal of biogenic carbon (CO<sub>2</sub>) incorporated in the product (mostly considering the corresponding uptake during biomass cultivation) and subsequent release at End of Life. However, the release pathway and occurrence of any storage was often not clearly specified (or not reported at all) for individual

end-of-life options. Dynamic accounting of biogenic carbon emissions and removals was conducted only in one study, considering the effects of their timing over a fixed 100-year time horizon. Beyond the deliberate choice of each study, this also reflects the absence of an established and agreed method in the scientific literature to conduct this type accounting, and the need to perform arbitrary choices for the time horizon to be applied in the assessment.

- Direct Land Use Change (dLUC) was explicitly addressed only in a few (eight) studies among those considering the use of primary bio-based feedstocks that may generate dLUC (i.e. 29 out of the 32 analysed in-depth) and in at least five other studies initially collected and screened for possible review. The associated effects were quantified either independently or as part of overall land use change effects incorporating explicitly or implicitly also the contribution of iLUC. Where relevant, the occurrence, extent and type of dLUC (e.g. share of cultivated land affected by dLUC, previous land use, etc.) were determined mainly based on assumptions and/or (region-specific) investigations and data, while a dedicated model was directly applied only in one case. In several cases, the assumptions inherently performed in the determination of the applied emission factors per unit of feedstock or land used for its cultivation were implicitly taken. No common approaches, models or factors were applied to quantify the corresponding GHG emissions, these being mostly based on different emission factors from the literature, or estimated according to the EC guidelines for the calculation of land carbon stocks within the context of the Renewable Energy Directive (EC, 2010). Despite a set of common rules has been set out in the PAS 2050 standard since 2011 (BSI, 2011), these were followed only in one study.
- Similarly to dLUC, indirect Land Use Change (iLUC) was explicitly addressed only in a few (seven) studies among those focusing on feedstocks with a risk of generating iLUC (i.e. 29 out of the 32 analysed in-depth) and in nine studies not included in the in-depth assessment. The effects from iLUC were quantified either independently or as part of overall land use change effects explicitly or implicitly including also the contribution of dLUC. Quantification was limited to potential GHG emissions, based in most cases on different emission factors from the literature or government-adopted default values (per unit of feedstock or land used for cultivation), or less frequently by directly applying specific casual-descriptive or economic equilibrium models. This reflected the current absence of a commonly accepted, consolidated and consistent method for iLUC accounting and quantification in a LCA context.
- The use of waste/residual bio-based feedstocks for plastics production was addressed through a variety of approaches and methods, ranging from a “cut-off” or “zero burden” approach applied to actual waste or wastewater streams, to different forms of allocation or system expansion for residues and by-products. This is expected to generate likely different results, although these were not analysed in detail for the reviewed studies.
- Similarly to the modelling of the use of waste/residual feedstocks, multi-functionality occurring at the biorefinery level due to generation of different co-products was not handled uniformly across the reviewed studies. Different approaches ranging from allocation based on different criteria or system expansion via substitution were applied, individually or in combination (e.g. distinguishing between energy and physical products). Again, this is likely to deliver different results, albeit the effects of applying different choices were not specifically investigated in the review.
- Additives were included, limited to production impacts, only in few studies (i.e. 2, not included in the in-depth assessment), mostly based on range of concentrations of relevant additive categories (i.e. not specific substances) and range of emission/impact factors for representative substances or combination of substances for each category (taken from the literature, existing Life Cycle Inventory databases or proprietary data). This reflected the scarcity of disclosed and accurate information

on additives use in plastic polymers and products, and of suitable specific inventory data applicable for modelling. Possible emissions and impacts occurring in other life cycle stages than production (e.g. during use or End of Life) were not quantified nor taken into account.

- Biodegradation of plastic products and materials during suitable biological end-of-life treatments such as composting and anaerobic digestion was modelled based on biodegradation rates generally varying over a wide range across the reviewed studies. These also applied different and sometimes contrasting approaches to quantify emissions from treatment (e.g. composting) and possible benefits from subsequent use of any resulting residual organic material (e.g. substitution of mineral fertilisers).
- Potential (indirect) consequences of the use of biodegradable plastic products, e.g. carrier bags reused for collection of organic waste and mulch film, were investigated in a few cases. The related modelling was based on different assumptions from dedicated studies or existing literature, and submitted to a critical review panel. However, while being sometimes conservative, such assumptions and evidence are not necessarily comprehensive, robust and certain enough for taking those aspects into account in a quantitative manner in a LCA study, and hence presenting a high risk of not ensuring a reliable, consistent and reproducible assessment.
- Littering and micro-plastics generation and impacts were generally neglected. Only one study semi-quantitatively addressed generation and impacts of product littering at End of Life, by approximating emissions from degradation in the terrestrial environment, and through the introduction of an indicator attempting to capture the aesthetic/visual impact of littered plastic products. This also reflects current incomplete knowledge of plastics degradation and transport pathways in the open environment and of the resulting emissions, as well as the absence of a comprehensive and solid method to address the potential impacts from such emissions.

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