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# Guide for interpreting life cycle assessment result

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## Abstract

Interpretation of the results of a life cycle assessment (LCA) study is a mandatory phase of LCA and it is a key aspect in order to derive robust conclusions and recommendations. One of the key aims of LCA is to provide the decision makers with comprehensive and understandable information: this task is achieved by a proper interpretation of the results of an LCA study.

The robust interpretation of a LCA study needs specific guidance to support the decision making process, both in policy and business contexts. Existing standards and guidelines provide a framework for interpretation, but there is the need of improving comprehensiveness and practical aspects thereof. This report aims at providing practitioners with a practical guidance on what are the aspects to be always taken into account when interpreting results of an LCA. Building on this report, further guidance needs to be developed to support interpretation of LCA studies both for LCA practitioners and LCA users (e.g. decision makers).

This report provides a practical and schematic workflow to be followed during interpretation, starting from the assessment of the hotspots of the baseline scenario, which is completed by sensitivity, completeness and consistency checks, to be performed at the level of the life cycle inventory (LCI) and life cycle impact assessment (LCIA). Additionally, case studies are reported to illustrate specific issues related to interpretation of results.

# 1 Introduction

Life cycle assessment (LCA) is more and more considered a reference method for the evaluation of supply chains, production and consumption systems, up to regions (Hellweg and Mila I Canals, 2014).

Life cycle interpretation is one of the four phases identified in the ISO 14040 and the ISO 14044 standards (ISO 2006a,b). In fact, according to the ISO standard interpretation of Life Cycle Inventory (LCI) or Life Cycle Impact Assessment (LCIA) phases according to the goal and scope of the study is a required step of an LCA study.

ISO 14044 further specifies that interpretation comprises the following elements: i) an identification of the significant issues based on the results of the LCI and LCIA phases of LCA; ii) an evaluation that considers completeness, sensitivity and consistency checks; iii) conclusions, limitations, and recommendations.

In 2013, the European Commission has put forward a communication which aims at harmonising the application of LCA for evaluation green products (Building the Single Market for Green Products, EC, 2013a) calculating their Environmental Footprint (EF) (both Product environmental footprint – PEF, and Organisation environmental footprint – OEF). The communication is accompanied by the PEF Guide (EC, 2013b) which explains that interpretation of the results of a PEF study serves two purposes: i) to ensure that the performance of the PEF model corresponds to the goals and quality requirements of the study; in this sense, PEF interpretation may inform iterative improvements of the PEF model until all goals and requirements are met; and ii) to derive robust conclusions and recommendations from the analysis, for example in support of environmental improvements.

The International Reference Life Cycle Data System (ILCD) Handbook (EC, 2010) clarifies that interpretation of an LCA has two main purposes that fundamentally differ: i) steering the work towards improving the Life Cycle Inventory model to meet the needs derived from the study goal; ii) especially for comparative LCA studies (while partly also applicable to other types of studies) it serves to derive robust conclusions and, often, recommendations.

Notwithstanding that several methodological guidance exists, on the different steps of LCA, so far, the interpretation phase has been little systematized, even though its importance is well recognized within the LCA community. LCA studies sometimes formulate conclusions and recommendations disregarding the uncertainties and the lack of consistency within the LCI and LCIA steps and across the goal and scope definition.

The need of guidance for the interpretation phase is becoming even more important as LCA is being increasingly recognised and used by various stakeholders. LCA has recently become a reference methodology for decision support in the policy context, in which its results may impact entire sectors or societies. In the EU context, these recent trends are reflected through initiatives and pilots related to the above-mentioned Building the Single Market for Green Products (EC, 2013a) as well as the new inclusion of LCA among the methods in the EU Better Regulation toolbox (EC, 2015) which identified tools relevant for conducting the impact assessment of policies. In this setting, robust and sound interpretation of LCA results is a must.

Several authors have provided evidence that interpretation is one of the key steps of an LCA study and give guidance and/or recommendations depending on the purpose of the study. Skone (2000) pointed out that the purpose of performing life cycle interpretation is to determine the level of confidence in the final results and communicate them in a fair, complete, and accurate manner. Initially, it was recognized that interpretation was not one of the hot topics in literature studies (Heijungs et al., 2001), and authors provided numerical techniques for interpretation. More recently other authors, such as Guadreault et al. (2009), while recognizing that LCA has become an important methodology for more sustainable process design, observed that its application in a decision-making context has been limited by a poor understanding of methodological

choices and assumptions, therefore they recommend careful interpretation of results to improve the quality of the outcome (i.e. improve the decision-making process). This view is shared by authors such as Prado Lopez et al. (2014), who have identified the lack of robust methods of interpretation to support decision makers, hence, they provide a novel approach based on a multi-criteria decision analytic method (stochastic multi-attribute analysis for life-cycle impact assessment (SMAA-LCIA)) which in their view should support both interpretation of results and policymakers. Van Hoof et al. (2013) explained how normalization helps maintain a multi-indicator approach while keeping the most relevant indicators, allowing effective decision making. Finally, other authors, such as Cellura et al. (2011) and Huang et al. (2013) performed LCA of specific products and they pointed out the relevance of sensitivity analysis to strengthen the reliability of the results obtained and draw conclusions to support sector specific guidelines. A structured approach covering the LCIA phase has been proposed (Castellani et al 2017), highlighting the importance of a systematic sensitivity analysis of impact assessment models, normalization and weighting set. Regarding normalisation and weighting steps, which are optional according to ISO standards, a recent overview of approaches, strengths and limitations has been provided in the context of activities of the UNEP/SETAC Life Cycle Initiative (UNEP/SETAC LCI) by Pizzol et al (2016).

The objective of this technical report is to provide guidance on how to perform an interpretation of LCI and LCIA results, based on what are the meaningful aspects to be taken into account, common to the majority of LCA studies.

This technical report aims to be a practical document to be used by LCA practitioners as a checklist of the main aspects to consider when interpreting LCI and LCIA results. Therefore, this report makes extensive use of examples, based on real case studies that may help critically assess LCA results, ensuring a reliable decision- and policy-making support.

A general framework on how to perform a proper interpretation is presented in chapter 2; the key aspects will be discussed further on in the report, using case studies as an example to guide the reader and the practitioner through "real life" applications.

Chapter 3 focuses on the identification of significant issues, namely the main contributors to the LCIA results (i.e. hotspots, most relevant impact categories, life cycle stages, processes and elementary flows).

Chapter 4 and 5 focus on the evaluations (sensitivity, completeness and consistency checks) in relation to the LCI and LCIA phases, respectively. Regarding the LCI, the report goes through the analysis of the data sources, by giving examples on how to interpret the system boundaries and the foreground system); it provides an overview on data quality and it comprises further completeness and consistency checks (e.g. cut-off, long term emissions, anomaly checks). When analysing the LCIA the report provides insights on sensitivity analysis regarding characterization, normalization and weighting. When dealing with characterization, the attention of the reader is drawn also on issues such as: the mapping between the inventory and the characterization models; and the evaluation of uncharacterized elementary flows.
Chapter 6 and 7 deal with overarching topics, such as estimation of uncertainty of the results of LCA study and meta-analysis.

# 2 A practical scheme to interpret LCI and LCIA results

The initial step of the interpretation phase is to review information from the first three phases of the LCA process, such as the ones used to build the study goals and scope, quality assurance procedures, reporting requirements, results, assumptions, external involvement (e.g. stakeholders, peer reviews) (Skone, 2000).

Both ISO 14044 and the ILCD Handbook (EC-JRC, 2011a) propose a scheme with the elements to be considered in the interpretation phase, and their relation to the other phases of the LCA study.

The elements can be grouped in:

- identification of significant issues (based on the results of the LCI and LCIA phases)

  the purpose of this first element of interpretation is to analyse and structure the results of earlier phases of the LCI/LCA study in order to identify the significant issues. There are two interrelated aspects of significant issues: i) firstly there are the main contributors to the LCIA results, i.e. most relevant life cycle stages, processes and elementary flows, and most relevant impact categories; ii) secondly, there are the main choices that have the potential to influence the precision of the final results of the LCA. These can be methodological choices, assumptions, foreground and background data used for deriving the process inventories, LCIA methods used for the impact assessment, as well as the optionally used normalisation and weighting factors.

- evaluation that considers completeness, sensitivity and consistency checks

  Completeness checks on the inventory are performed in order to determine the degree to which it is complete and whether the cut-off criteria have been met. Sensitivity checks have the purpose to assess the reliability of the final results and of the conclusions and recommendations of the LCA study. The consistency check is performed to investigate whether the assumptions, methods and data have been applied consistently throughout the LCI/LCA study.

- conclusions, limitations and recommendations.

  Integrating the outcome of the other elements of the interpretation phase, and drawing on the main findings from the earlier phases of the LCA, the final element of the interpretation is to draw conclusions and identify limitations of the LCA, and to develop recommendations for the intended audience in accordance with the goal definition and the intended applications of the results.

In this report, a schematic workflow representing a practical approach to interpret LCI and LCIA results is proposed in Fig.1, building on both ISO 14044 (ch. 4.5.1.1) and the ILCD Handbook (ch. 9.1) (EC-JRC, 2011a). The workflow is to be seen as an iterative approach to finally draw robust conclusions and recommendations.

The scheme should not be seen as substituting the approaches of the ISO 14044 and the ILCD Handbook, instead it should be seen as a practical application of the requirements laid down in these documents.

Furthermore, the scheme should not be intended as a comprehensive list of items to check, instead it should be seen as a normal workflow that should be carried out by an LCA practitioner in order to derive robust conclusions out of the LCA study. More accurate and case-specific checks should be done depending on the intended goal and scope of the study.

Further aspects, such as the appropriateness of the functional unit, the intended goal, the system boundaries and value choices should normally be part of an interpretation, in order to identify recommendations and limitations of the study and draw the appropriate conclusions. They are not discussed further in this report, but it is recommended that at

least, each decision based on a personal or a stakeholder's value shall always be documented and explained in the conclusions of the LCA study. This is consistent with one of the key aims of LCA, which is to provide the decision makers with comprehensive and understandable information in a transparent manner.

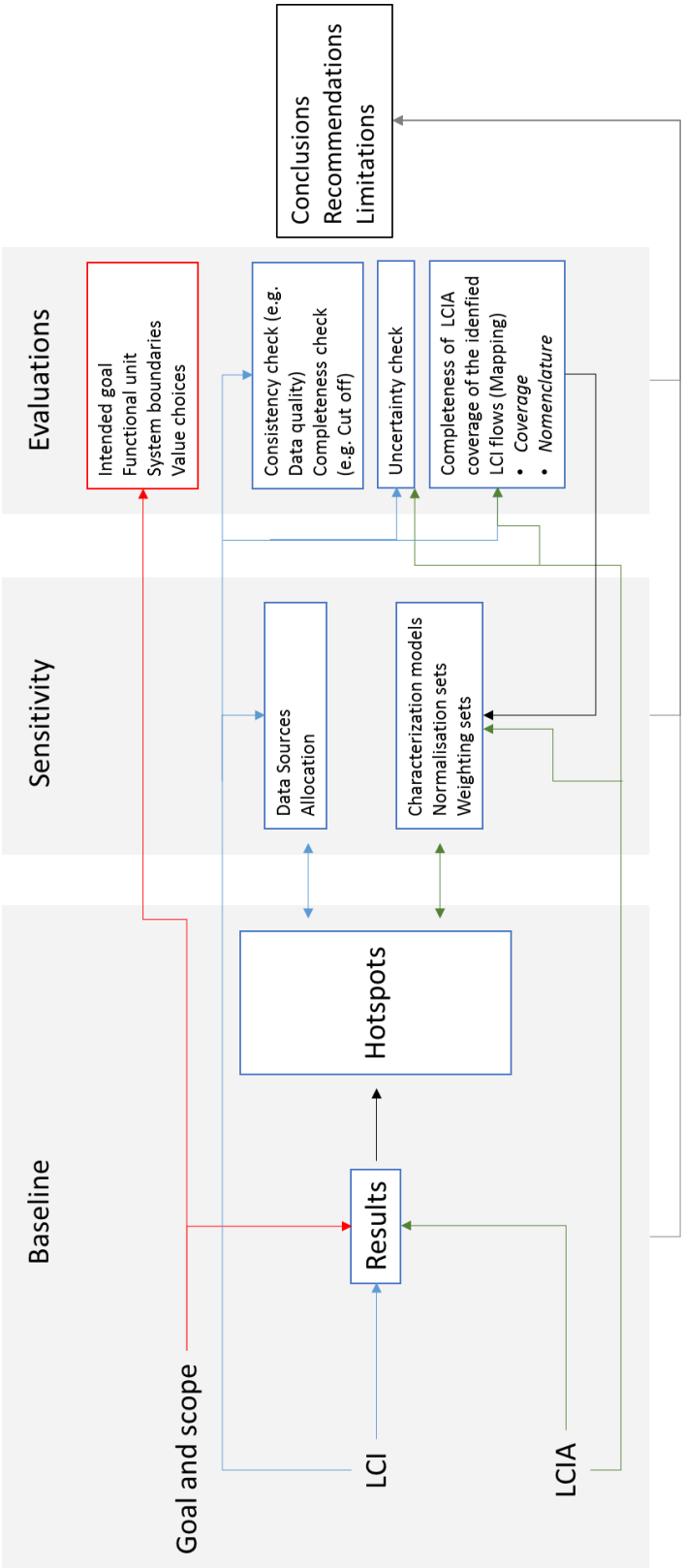


*Figure 1 Practical application of requirements in ISO 14044 and ILCD Handbook to interpret LCI and LCIA results.*

# 3 Identification of key issues: Hotspots analysis

In order to get started with interpretation of results, it is necessary to identify what the key issues are. Once the key issues are identified it is possible to further evaluate the overall robustness of the LCA study by using for example completeness, consistency and sensitivity checks.

This chapter on hotspot analysis gives an overview of the overarching methodological framework of hotspot analysis as currently developing with the UNEP/SETAC Life Cycle Initiative - Flagship Project 3a and the hotspot analysis as tested in the Environmental Footprint (EF) pilot phase (2013 – 2017). In the first case, hotspot analysis is regarded as a subject on its own, in the second case it is shown a practical way to perform a hotspot analysis in an environmental footprinting context: in the EU Environmental Footprint, it is more focused on the interpretation of the screening study results of the average or representative product[1]. The hotspots in a EU Environmental Footprint may be, but does not have to, found in a contribution analysis of the characterized results.

Therefore, in this chapter, we first describe the hotspot analysis as outlaid in the UNEP/SETAC LCI and then the hotspot analysis as performed in the EF pilot phase.

## 3.1 The hotspot analysis in the UNEP/SETAC Life Cycle Initiative

The UNEP/SETAC Life Cycle Initiative has been running a Flagship project (nr. 3a) on hotspot analysis, since 2012 until 2017. According to UNEP/SETAC (2016), the benefits of hotspots analysis include ensuring:

- Focus on priority issues (e.g., waste, water, materials of concern)
- Focus on the right life cycle stage (e.g., material acquisition, manufacturing, use, end of life)
- Focus on the right actors (e.g. producers, manufactures, suppliers, retailers, customers) to evaluate, influence and implement solutions
- Implications of trade-offs are understood
- Resources (e.g. time, money) can be effectively allocated to actions.

Previously, there was not currently a common global approach to hotspots analysis; nor has there been sufficient effort to bring together or share best practice amongst those organisations or initiatives currently developing and using these methods. Nor was there any accepted guidance on how to translate and apply the results of hotspots analysis into meaningful sustainability information and insight for use by industry, governments and other stakeholders. Therefore, the UNEP SETAC Life Cycle Initiative established a new Flagship Project to address these and other issues. The objectives of the UNEP/SETAC LCI Flagship Project 3 are to produce:

1. A common methodological framework and global guidance for sustainability hotspots analysis;
2. A protocol for the appropriate use and communication of sustainability information derived from hotspots analysis;
3. To evaluate and, if possible, implement a range of options to bring together the findings from existing hotspots studies to provide a richer, global picture of sustainability hotspots in the economy and society.

Phase 1 was finished in December 2014 and gives an overview of different tools and methods that can be used in the hotspot analysis. This includes 42 hotspots analysis

[1] In the context of the EF pilot phase, within each Product Environmental Footprint Category Rule (PEFCR) pilot a "representative product" has to be defined, which is representing >50% of the European market. The PEF screening study is a PEF study conducted on the representative product: one of the aims of the screening study is to identify the hotspots, most relevant elementary flows, processes and life cycle stages.

methodologies divided into product, sector and national scale of application. Of these, 21 methodologies (4 at the national-level; 5 at the sector-level; and 12 predominantly applied at the product category-level) were shortlisted and further analysed. Examples of methods are *EU EIPRO* and *Global Protocol for Community Scale GHG emissions* on the national level, *GHG Protocol Corporate Value Chain (Scope 3) Accounting and Reporting Standard* and *Sustainability Accounting Standards Board Materiality Map™* on the sector level and *French Grenelle I and II BPX 30-323-0 Product Lifecycle Environmental Impact Quantification Guidance Standard* and *Water Quality Association (WQA) Hotspots Analysis* on the product level. Please refer to Figure ES 2 in (Barthel et al 2014) for the complete list.

This first phase of the project fed into the second phase of the Flagship project.

Phase 2 is running from January 2015 – January 2017 and produces the methodological framework (UNEP/SETAC, 2016) and the communication guidance[2]. These are currently in a six-weeks public consultation (27. Oct – 15 Dec 2016). The framework provides:

- a recognized set of guiding principles and practices for hotspots analysis
- a globally agreed methodological framework for hotspots analysis
- agreement on the appropriate use and communication of product sustainability information derived from hotspots analysis

Future step will be to work with hotspot analysis for cities and regions, and road testing of the methodological framework.

## 3.2 The hotspot analysis in the Environmental Footprint pilot phase (2013 – 2017)

The 2013-2017 Environmental Footprint (EF) pilot phase has three main objectives:

- test the process for developing product- and sector-specific rules;
- test different approaches to verification;
- test communication vehicles for communicating life cycle environmental performance to business partners, consumers and other company stakeholders.

This is tested by groups of organisations called Technical Secretariats (TS), who volunteered to develop the rules for their product or sector.

The hotspot analysis as described below were tested and developed in an iterative way on the approximatively 25 pilots participating in the EF pilot phase to develop Product Environmental Footprint (PEF) Category Rules (PEFCR) or Organisation Environmental Footprint (OEF) Sector Rules (OEFSR).

A hotspot analysis is required for the PEF/OEF screening studies of the average or representative product for the PEFCR/OEFSR and for the supporting studies[3] (EF study of a single product/organisation belonging to the product category/sector) when developing PEFCR/OEFSR (PEFCR Guidance document, (EC, 2016)).

The hotspot analysis in the EF context aims to identify the hotspots, the most relevant impact categories, life cycle stages, processes and elementary flows, as shown in Fig 2 and it provides a guidance on how to do that.

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[2] *Communicating hotspots: The effective use of sustainability information to drive action and improve performance* from http://www.wrap.org.uk/sites/files/wrap/Hotspots%20Communications%20Guidelines%20v.3.5%207Oct2016.pptx

[3] The EF supporting studies are PEF/OEF studies run by single companies, with the aim of testing the draft PEFCRs and OEFSRs.

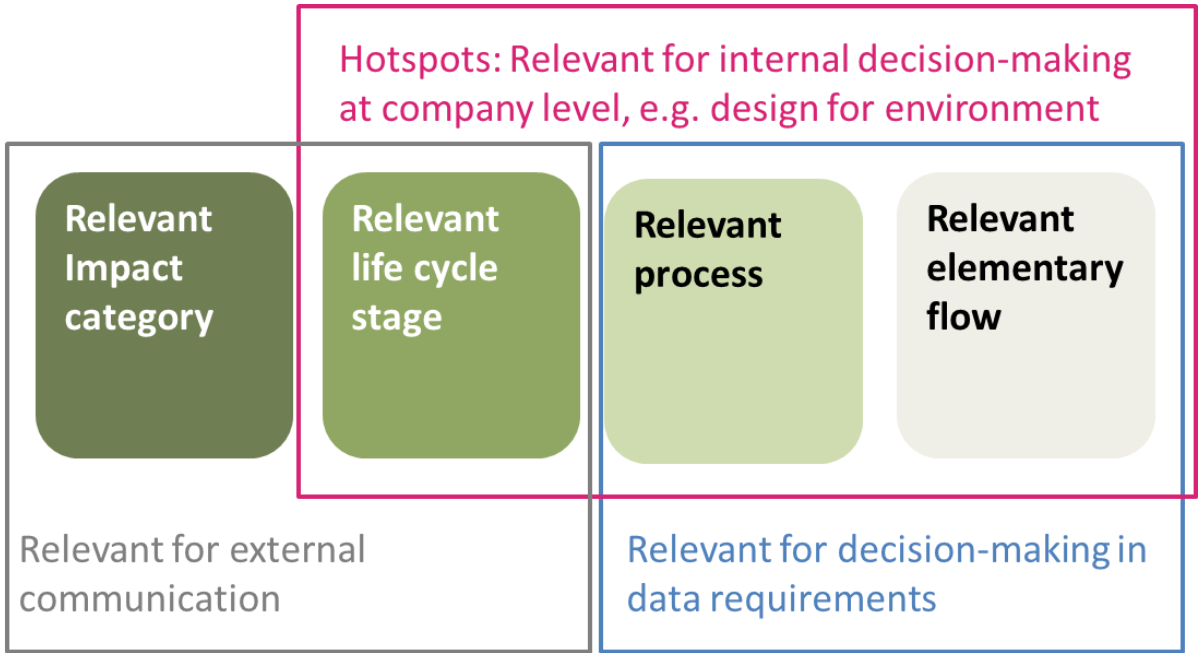


*Figure 2: Hotspot analysis in the context of the Environmental Footprint pilot (James & Galatola, 2015)*

As shown in the figure, the hotspots (and thereby also the hotspot analysis) is important for the decision on which life cycle stages, processes and elementary flows that are hotspots (> 50 % contribution) and which are relevant (> 80 % contribution).

The hotspots might serve the purpose of "warning" an organisation about the area where they should focus their attention in order to improve the environmental performance of a product (PEF) or an organisation (OEF). Whilst traditionally LCA practitioners are expected to provide primary/site-specific data for processes taking place in the foreground and secondary data for processes taking place in the background, in PEF/OEF the implementation of the materiality principle leads to a different approach. The processes most contributing to the final results (in terms of environmental impacts or savings) are those for which the best quality of data should be provided. For the processes less relevant it is possible to use data of lower quality.

### 3.2.1 Procedure to identify the most relevant impact categories

Starting from the normalised and weighted results of PEF screening study, each Technical Secretariat is asked to select the impact categories that they consider more relevant in terms of "communication purposes".

### 3.2.2 Procedure to identify the most relevant life cycle stages

The proposed guideline by the PEFCR Guidance (EC, 2016) is to consider as relevant all life cycle stages which together contribute over 80% (before normalisation and weighting) to any of the baseline impact categories. This should start from the largest to the smallest contributions. In addition, more life cycle stages can be added to the list of the most relevant ones.

In order to guarantee a minimum level of harmonisation among different PEFCRs/OEFSRs, a list of default life cycle stages was identified:

- Raw material acquisition and pre-processing (including production of parts and unspecific components);

- Production of the main product[4]
- Product distribution and storage
- Use stage scenario (if in scope);
- End-of-life (including product / part reuse, recovery / recycling; if in scope).

### 3.2.3 Procedure to identify the most relevant processes

Each impact category shall be investigated to identify the most relevant processes. The identification of the most relevant processes shall be done at the whole life cycle level. Similar/identical processes taking place in different life cycle stages (e.g. transportation) shall be accounted separately.

In the context of the EF pilot phase processes were defined as most relevant where they collectively contributed at least 80% to any impact category before normalisation and weighting.

In some instances, vertically aggregated datasets may be identified as representing relevant processes. It may not be obvious which process within an aggregated dataset is responsible for contributing to an impact category. The metadata accompanying the data should be reviewed and used to identify relevant processes. If this is not possible, it can be decided whether to seek further disaggregated data or to treat the aggregated dataset as a process for the purposes of identifying relevance[5].

### 3.2.4 Procedure to identify the most relevant elementary flows

As far as the most relevant elementary flows is concerned they are the ones contributing cumulatively more than 80% to the impact category. It can be decided whether this is calculated at the level of overall life cycle and / or single (most relevant) process level. All elementary flows contributing more than 5% to the impact category shall be considered as relevant.

Once the most relevant elementary flows have been identified they shall be linked to the processes emitting them.

### 3.2.5 Procedure to identify the hotspots

A hotspot can be identified at different levels of granularity: impact category, life cycle stage, process or elementary flow.

In the context of PEF/OEF pilot phase a hotspot is defined as either (James and Galatola, 2015):

- OPTION A: (1) life cycle stages, (2) processes and (3) elementary flows cumulatively contributing at **least 50%** to any impact category before normalisation and weighting (from the most contributing in descending order).
- OPTION B: At least the two most relevant life cycle stages, processes and at least two elementary flows (minimum 6). Additional hotspots may be identified by the TS.

[4] If it is not possible to differentiate between the production of the main product and ancillary materials and other life cycle stages, e.g. raw material acquisition and transport, then these life cycle stages can be merged. Adequate justifications shall be provided in the screening report and be highlighted in the consultation phase. If an "aggregated" life cycle stage/process is most relevant it implies that all the processes included will be classified as most relevant.

[5] In this last case, if an aggregated dataset is relevant, everything in it is automatically relevant.

### 3.2.6 Dealing with negative numbers

When identifying the percentage contribution from any process or flow it is important that absolute values are used (i.e. the minus sign is ignored) in calculating percentage contributions. This allows indentifying the relevance of any credits.

### 3.2.7 Specific instructions about aggregating elementary flows

The toxicity-related impact categories ("Human toxicity, cancer effects", "Human toxicity, non-cancer effects", and "Freshwater ecotoxicity") contain long lists of characterization factors specified on the elementary flow category, level 3 (also known as sub-compartment – for example urban air, freshwater, agricultural soil). The impact category "Particulate matter" also contains characterization factors at the level 3 category.

It is recommended to aggregate the elementary flows on the level 2 category (i.e. emissions to air, water, soil) in the contribution analysis. This is to make the analysis more manageable and because the level 3 category does not give additional information when analyzing on the process level (the processes already contain information on the location type of the emissions).

It is important to model foreground data on the level 3 category for particulates and toxic substance flows, because the characterization factors can deviate considerably.

Metal resource flows are not specified per origin of ore type in the source files of the ILCD recommended methods. However, in several background databases, metal resource flows are differentiated (for example, Silver, Ag 4.6E-5%, Au 1.3E-4%, in ore, Silver, Ag 4.2E-3%, Au 1.1E-4%, in ore, Silver, Ag 2.1E-4%, Au 2.1E-4%, in ore, etc.). Therefore, the specified flows were added to the ILCD method in LCA software packages with the same characterization factors as for the unspecified metals. When doing a contribution analysis of the metal resource flows, it is therefore recommended to aggregate the flows per metal (silver, copper, nickel, etc.).

There are 5 different fossil fuel related resource flows specified in the source files of the ILCD recommended methods (brown coal; 11.9 MJ/kg, crude oil; 42.3 MJ/kg, hard coal; 26.3 MJ/kg, natural gas; 44.1 MJ/kg, peat; 8.4 MJ/kg). However, in several background databases, fossil resource flows are specified with different calorific values (for example, Gas, natural, 46.8 MJ per kg, Gas, natural, 36.6 MJ per $m^3$, Gas, natural, 35 MJ per $m^3$, Gas, natural, 30.3 MJ per kg, etc.). Therefore, the specified flows were added to the ILCD method in LCA software packages with characterization factors related to the factors in the original source, taking the different calorific value into account. When doing a contribution analysis of the fossil resource flows, it is therefore recommended to aggregate the flows based on the 5 original flows (brown coal, crude oil, hard coal, natural gas and peat).

### 3.2.8 Conclusions

In table 1 the requirements to define most relevant contributions during the EF pilot phase are summarized as found in the PEFCR Guidance (EC, 2016).

*Table 1 Summary of requirements to define most relevant contributions and hotspots (James & Galatola, 2015)*

| **Item** | **At what level does relevance need to be identified?** | **Threshold** |
|---|---|---|
| Most relevant impact categories | In the final results, starting from normalized and weighted results but deviations possible if justified | No threshold. Decision left to TS but subject to stakeholder consultation and TAB opinion |
| Most relevant life cycle stages | For each impact category, before normalization and weighting. Not relevant for data needs identification | All life cycle stages contributing cumulatively more than 80% to any impact category |
| Hotspots | For each impact category, before normalization and weighting | Either (i) life cycle stages, processes, and elementary flows cumulatively contributing at least 50% to any impact category, or<br>(ii) at least the two most relevant impact categories, life cycle stages, processes and at least two elementary flows (minimum 6). Additional hotspots may be identified by the TS |
| Most relevant processes | For each impact category, before normalization and weighting. Essential for data needs identification | All processes contributing cumulatively more than 80% to any impact category |
| Most relevant elementary flows | For each impact category, before normalization and weighting. Essential for data needs identification | All elementary flows contributing cumulatively more than 80% to any impact category and in any case all those contributing more than 5% individually |

When assessing what is most relevant it is also recommended to check if what is identified as being non-relevant can be actually considered as such. A practical example is given at paragraph 4.3.1.

### 3.2.9 Example to identify most relevant contributions and hotspots

The PEFCR Guidance (2016) also provided an example on how to perform a hotspot analysis and the identification of most relevant impact categories, life cycle stages, processes and elementary flows. The example is reported here: it is a fictitious example, not based on any specific PEF study results.

#### 3.2.9.1 Example: most relevant Impact Categories

Table 2 shows the contribution of different impact categories based on normalised and weighted results

*Table 2 Selection of most relevant Impact Categories (James & Galatola, 2015)*

| Impact category | Unit | Contribution (%) |
|---|---|---|
| **Climate change** | kg $CO_2$ eq | 21.5 |
| **Ozone depletion** | kg CFC-11 eq | 3.0 |
| **Human toxicity, cancer effects** | CTUh | 8.3 |
| **Human toxicity, non-cancer effects** | CTUh | 14.9 |
| **Particulate matter** | kg $PM_{2.5}$ eq | 0.1 |
| **Ionizing radiation, human health** | Kbq $U_{235}$ eq | 0.5 |
| **Photochemical ozone formation** | kg NMVOC eq | 2.4 |
| **Acidification** | molc $H^+$ eq | 1.5 |
| **Terrestrial eutrophication** | molc N eq | 1.0 |
| **Freshwater eutrophication** | kg P eq | 1.0 |
| **Marine eutrophication** | kg N eq | 0.1 |
| **Freshwater ecotoxicity** | CTUe | 0.1 |
| **Land use** | kg deficit | 14.3 |
| **Water resource depletion** | $m^3$ water eq | 18.6 |
| **Mineral, fossil resource depletion** | kg Sb eq | 12.7 |

Based on the normalised and weighted results the TS can decide that the following impact categories are relevant for "communication purposes": climate change, water depletion and land use. Where there is deviation from the most significant contributors to the normalised and equally weighted results, justification shall be provided.

Once the relevant impact categories for communication purposes have been selected, the TS shall start identifying the most relevant life cycle stage, processes and flows per each impact category (all baseline EF impact categories, not only those relevant for communication purposes).

#### 3.2.9.2 Example: most relevant Life Cycle Stages

Table 3 shows the contribution of different life cycle stages to the climate change impact category (based on the characterised inventory results before normalisation and weighting).

*Table 3 Selection of most relevant life cycle stages (James & Galatola, 2015)*

| Life cycle stage | Contribution (%) |
|---|---|
| Raw material acquisition and pre-processing | 42.1 |
| Production of the main product | 25.2 |
| Product distribution and storage | 16.4 |
| Use stage (if in scope) | 10.8 |
| End-of-life | 5.5 |

The three life cycle stages in purple will be the ones identified as "most relevant" for climate change as they are contributing to more than 80%. This procedure shall be repeated for all the baseline EF impact categories.

#### 3.2.9.3 Example: most relevant Processes

Table 4 shows the contribution of different processes to the climate change impact category (based on the characterised inventory results before normalisation and weighting).

*Table 4 Selection of most relevant processes (James & Galatola, 2015)*

| Unit process | Contribution (%) |
|---|---|
| **Process A** | 8.9 |
| **Process B** | 61.4 |
| **Process C** | 23.4 |
| **Process D** | 2.8 |
| **Process E** | 1.5 |
| **Process F** | 0.9 |
| **Other processes** | 0.9 |

According to the proposed procedure the processes in orange shall be selected as "most relevant". Process E could be added to the as most relevant ones based on specific considerations (e.g. it is under operational control, it is of relevance for the specific sector, etc.). This procedure shall be repeated for all the baseline EF impact categories.

#### 3.2.9.4 Example: most relevant Elementary Flows

Considering that the selection of the most relevant elementary flows can be done at overall life cycle level (option 1) and/or per relevant process (option 2), there are two

possible outcomes. Starting from the inventory results provided in Table 5, the list of most relevant flows are highlighted in blue in Table 6 (at overall life cycle) and Table 7 (at process level).

*Table 5 Inventory results (climate change, results expressed in $gCO_{eq}$). (James & Galatola, 2015)*

| Inventory flow | Substance 1 | Substance 2 | Substance 3 | Substance 4 | Substance 5 | Total |
|---|---|---|---|---|---|---|
| **Process A** | 249 | 85 | 6 | 45 | 5 | 390 |
| **Process B** | 1100 | 600 | 500 | 450 | 50 | 2700 |
| **Process C** | 300 | 250 | 20 | 30 | 430 | 1030 |
| **Process D** | 60 | 30 | 20 | 10 | 5 | 125 |
| **Process E** | 64 | 1 | 1 | 1 | 1 | 68 |
| **Process F** | 15 | 10 | 8 | 5 | 3 | 41 |
| **Other processes** | 15 | 10 | 8 | 5 | 3 | 41 |
| **Total** | 1803 | 986 | 563 | 546 | 497 | 4395 |

*Table 6 Most relevant inventory flows contributing to climate change (based on the inventory results before normalisation and weighting) - overall life cycle (option 1). (James & Galatola, 2015)*

| Inventory flow | Substance 1 | Substance 2 | Substance 3 | Substance 4 | Substance 5 | Total |
|---|---|---|---|---|---|---|
| **Process A** | 5.7% | 1.9% | 0.1% | 1.0% | 0.1% | 8.9% |
| **Process B** | 25.0% | 13.7% | 11.4% | 10.2% | 1.1% | 61.4% |
| **Process C** | 6.8% | 5.7% | 0.5% | 0.7% | 9.8% | 23.4% |
| **Process D** | 1.4% | 0.7% | 0.5% | 0.2% | 0.1% | 2.8% |
| **Process E** | 1.5% | 0.0% | 0.0% | 0.0% | 0.0% | 1.5% |
| **Process F** | 0.3% | 0.2% | 0.2% | 0.1% | 0.1% | 0.9% |
| **Other processes** | 0.3% | 0.2% | 0.2% | 0.1% | 0.1% | 0.9% |
| **Total** | 41.0% | 22.4% | 12.8% | 12.4% | 11.3% | |

*Table 7 Most relevant inventory flows contributing to climate change (based on the inventory results before normalisation and weighting) – process level (option 2). (James & Galatola, 2015)*

| Inventory flow | Substance 1 | Substance 2 | Substance 3 | Substance 4 | Substance 5 | Total |
|---|---|---|---|---|---|---|
| **Process A** | 64% | 22% | 2% | 12% | 1% | 100% |
| **Process B** | 41% | 22% | 19% | 17% | 2% | 100% |
| **Process C** | 29% | 24% | 2% | 3% | 42% | 100% |

| | | | | | | |
|---|---|---|---|---|---|---|
| **Process D** | 48% | 24% | 16% | 8% | 4% | 100% |
| **Process E** | 94% | 1% | 1% | 1% | 1% | 100% |
| **Process F** | 37% | 24% | 20% | 12% | 7% | 100% |
| **Other processes** | 37% | 24% | 20% | 12% | 7% | 100% |

### 3.2.9.5 Example: hotspots

In Tables 8 and 9 the identified hotspots - based on the two different approaches available- are presented.

*Table 8 Hotspots based on 50% cumulative contribution (OPTION A). (James & Galatola, 2015)*

| | **Hotspots** |
|---|---|
| **Life cycle stages** | • Raw material acquisition and pre-processing<br>• Production of the main product |
| **Processes** | • Process B |
| **Elementary flows** | Option 1: Substance 1 and 2<br>Option 2:<br>• Substance 1 and 2 in processes B<br>• Substance 1 and 5 in process C<br>• Substance 1 in process E |

*Table 9 Hotspots based on top two representatives (OPTION B). (James & Galatola, 2015)*

| | **Hotspots** |
|---|---|
| **Life cycle stages** | • Raw material acquisition and pre-processing<br>• Production of the main product |
| **Processes** | • Process B<br>• Process C |
| **Elementary flows** | Option 1: Substance 1 and 2 in process B<br>Option 2: Substance 1 in process E[6] and substance 5 in process C |

# 4 Interpretation on Life Cycle Inventory (LCI) level

During the life cycle inventory phase, the actual data collection and modelling of the system has to be done, in line with the goal definition and aiming at meeting the requirements derived in the scope phase. The LCI results are the input to the subsequent Life Cycle Impact Assessment (LCIA) phase.

Interpretation involves evaluations at the level of Life Cycle Inventory, in order to:

- improve the inventory model to meet the needs derived from the study goal,
- perform a sensitivity analysis to check for limitations in the appropriateness of the life cycle inventory work,

[6] Choosing the hotspots based on option B-2 might lead to anomalies (like in this fictitious example) where an elementary flow is identified as hotspot even if its absolute value is not prominent.

- understanding the underlying assumptions when secondary datasets are used in the model,
- combining the above points to refine the LCA model,
- draw appropriate and robust conclusions.

Key aspects to evaluate life cycle inventories are the consistency and completeness thereof. They can be evaluated taking into account different aspects. A sensitivity check to evaluate the reliability of the results is also a needed step.

A first aspect to be taken into account is the data source. Primary data are collected directly by the company performing the study, thus the consistency of the data used with the goal and scope of the study should be more straightforward. However, when and LCA study is making use of secondary datasets to model the background system, a careful analysis of the consistency of these datasets with the primary data collected, and in relation with the goal and scope, shall be made.

Indeed, different modelling assumptions in datasets aimed at representing the same product system can lead to different results, affecting the reliability of the LCA study (Williams et al., 2009). An example of the relevance of different data sources and consistency has been discussed in Corrado et al (2017): this study focuses on different data sources to originate the inventory of arable crops (1 kg of wheat). The choice of the secondary dataset can have a relevant influence on LCA results and it is considered among the challenges for a robust LCA (Notarnicola et al. 2017a).

A second aspect to be taken into account regarding life cycle inventories is the quality of the data used, regarding time, geographical and technological representativeness, precision, methodological compliance and documentation. Therefore, in this chapter a paragraph is dedicated to analyzing more in detail this aspect.

A third relevant aspect in relation to LCIs is the consistency and completeness in relation to single aspects, such as cut-off, inclusion/exclusion of specific elementary flows (e.g. long term emissions). It is not straightforward to judge if an inventory can be considered complete. As a general rule, it has always to be kept in mind that the inventory has to be checked in its completeness in relation to the goal and scope of the study: for example, if the goal of the study is to assess only the climate change impact category, the completeness of the inventory shall be evaluated against those emissions that are actually contributing to climate change within the system boundaries. It does not lead to improved conclusions the compilation of a more detailed inventory, with the inclusion of e.g. substances that do not contribute to climate change.

## 4.1 Analysis of data sources

When selecting secondary data sets, it is important to ensure that all data sets used in the modelling of the system model are methodologically consistent (i.e. consistency check). The use of inconsistent data can lead to an unreliable and distorted LCA study, often with wrong conclusion and recommendations drawn. Furthermore, use of datasets with different level of completeness (i.e. completeness checks) may also lead to unreliable conclusions.

The study from Corrado et al. (2017) is used as a case study to highlight the relevance of different data sources in an LCA study. Therefore, the content of this paragraph is mainly based on this paper. The paper is organized as follows: firstly, the authors report an overview of the system boundaries and underlying assumptions adopted in building secondary datasets for arable crops within three databases. Secondly, they summarise the approaches adopted to model the foreground system, highlighting analogies and differences. Afterwards, they point out the influence that modelling approaches can have on the impact assessment. Finally, the combination of these elements, allowed to derive some considerations on relevant elements of datasets.

It is focused on the analysis of secondary datasets for arable crops production, as modelled in three databases adopted as source of secondary data in LCA: Agribalyse v 1.2 (Colomb et al., 2015), Agrifootprint v 1.0 (Blonk Agri-footprint BV, 2014) and Ecoinvent v 3.1 (Weidema et al., 2013). For simplicity, in the remainder of the report they are named Agribalyse, Agrifootprint and Ecoinvent respectively.

The authors performed three types of analysis:

- comparison of the system boundaries and underlying assumptions;
- analysis of how the foreground system is modelled in each dataset, focusing on different aspects, such as: fertilisers application and nutrient fate; plant protection products application and fate; heavy metals (HMs) input, mass balance and fate; irrigation; agricultural operations; land occupation and transformation;

  comparison of the results of the life cycle impact assessment for the foreground system, including the relative contribution of background.

### 4.1.1 Analysis of the system boundaries

The system boundaries and the main underlying assumptions adopted to model arable crops production in the three databases are reported in Table 10.

*Table 10 Underlying assumptions in relation to the three databases considered by the authors (Corrado et al., 2017)*

| | | **Agribalyse** | **Agrifootprint** | **Ecoinvent** |
|---|---|---|---|---|
| **Data source** | | Provided by technical institutes (e.g. ARVALIS – Institut du Végétal) | Different sources (e.g. scientific literature, official statistics such as FAOstat, Eurostat) | GL-Pro project – Barrois region (Nemecek and Baumgartner, 2006) |
| **Straw management (when applicable)** | | Partly removed from the field | Totally removed from the field | Left on the field |
| **Allocation of co-products (grains and straw)** | | Not applied because the straw market was not very structured at the time when the datasets were developed | Economic, mass and energy allocation | Not applicable because straw is assumed to be left on the field |
| **Nutrients from straw left on the field** | | The fertilising effect of crop residues and emissions from the residues are allocated to the crop that generated the residues | Not applicable | The fertilising effects of crop residues are allocated to the crop that generated it (only for P and K). The amount of fertilisers is corrected for the amount of nutrients in crop residues. |
| **Crop rotation modelling** | **Phosphorous (P) and potassium (K) inputs and emissions allocation** | P and K fertilisers production and emissions due to their application are allocated to each crop pro rata for the crop exports | Not reported | The amount of P and K of the residues supplied to the field through residues were allocated to the crop that generates them |

| | | | | |
|---|---|---|---|---|
| | **N inputs and emissions allocation** | The organic nitrogen available for the crop to which the fertiliser is applied, is allocated to that crop. The remaining fraction that contributes to increasing the stock of organic matter was allocated to all the crops in the rotation. Mineral nitrogen is completely allocated to the crop to which it was applied. | Not reported | Allocation not performed |

### 4.1.2 Foreground system analysis

The authors compared the activities modelled in the foreground system of the datasets referring to seven field activities: agricultural operations, fertilisers application and nutrients environmental fate, plant protection products (PPP) application and related environmental fate, heavy metals (HMs) input and related environmental fate, irrigation and land occupation and transformation. The average crop yields reported in the dataset presents small differences (wheat min-max; barley min-max, rapeseed min-max; pea min-max). The 95% confidence interval of the yields is defined only in Agrifootprint datasets, whereas it is not reported in Agribalyse and Ecoinvent datasets.

For completeness, findings and aspects to be taken into account are below reported. The below checklist can also be used to build a completeness check of the different datasets and it can be used to build a sensitivity analysis based on the parameters that are seen to be affecting the final results.

<u>Agricultural practices modelling</u>

- Management of agricultural residues
- Composition of the crop rotation
- Allocation to the co-products

<u>Agricultural operations modelling</u>

- Number of operations
- Number of passages
- Machine power
- Soil texture
- Airborne emissions due to fuel combustion
- Emissions of heavy metals due to tires abrasion

<u>Fertilisers application and nutrients fate modelling</u>

- The nutrients fate is highly influenced by site-specific conditions
    - environmental conditions,
    - soil type,
    - agricultural management practices
    - type of fertiliser (Brentrup et al., 2000)
    - spatial differentiated emissions when modelling agricultural systems
- Nomenclature (e.g. the paper shows that phosphorus compounds emissions were expressed in the datasets with different flows, limiting the possibility of making a comparison among inventory data).

<u>PPP application and environmental fate modelling</u>

- application method,
- weather and soil conditions,
- crop characteristics
- irrigation
- active substances whose use is not anymore authorised

Heavy metals inputs and environmental fate modelling

- Mass balance
- Fate

Irrigation modelling

- Irrigation volumes
- Water flows type
- Environmental conditions
- Spatially-differentiated water flows

Land use transformation modelling

- Amounts of transformed land are considered.
- Type of land transformed (e.g. from … to …)
- $CO_2$ emissions due to land transformation

## 4.2 Data quality

As data is essential in all quantitative studies like LCAs, so is also their quality. Therefore, the data applied in the life cycle inventory phase shall be assessed in the interpretation phase against the goal and scope of the study: is the data quality sufficient for the goal of the study? Is the data consistent? Can the data be seen as complete?

If the answer is yes, then the interpretation can be preceded and conclusions drawn. If no, then either the goal and scope need to be adjusted, or probably in most cases better, the life cycle inventory phase need to be revisited in an iterative manner to improve the data quality to the appropriate level. The assessment of the data quality is a mandatory reporting element in the interpretation phase.

ISO 14044 (2006) defines *data quality* as "characteristics of data that relate to their ability to satisfy stated requirements". The data quality requirements are described in the goal and scope of the LCA study and covers the following 10 aspects (ISO14044, 2006, ch. 4.2.3.6):

1) time-related coverage: age of data and the minimum length of time over which data should be collected;
2) geographical coverage: geographical area from which data for unit processes should be collected to satisfy the goal of the study;
3) technology coverage: specific technology or technology mix;
4) precision: measure of the variability of the data values for each data expressed (e.g. variance);
5) completeness: percentage of flow that is measured or estimated;
6) representativeness: qualitative assessment of the degree to which the data set reflects the true population of interest (i.e. geographical coverage, time period and technology coverage);
7) consistency: qualitative assessment of whether the study methodology is applied uniformly to the various components of the analysis;

8) reproducibility: qualitative assessment of the extent to which information about the methodology and data values would allow an independent practitioner to reproduce the results reported in the study;

9) sources of the data;

10) uncertainty of the information (e.g. data, models and assumptions).

In the EF methods[7], the consistency and reproducibility is one of the main aims of any EF study and not evaluated as part of the data quality assessment. Data sources should be ILCD/EF compliant and specified if a PEFCR/OEFSR is followed. Precision and uncertainty are evaluated together in the EF method. Therefore, the EF operates with six semi qualitative data quality requirements instead of 9 in ISO14044. (cf. section 5.6 in European Commission, 2013).

In order to evaluate the appropriateness and the quality of the data used to compile the LCI, they have to be scored against certain requirements. As an example the table below (Table 11) provides the data quality requirements for a PEF study of intermediate paper products (EC 2013a, Schau and Davidsson 2015).

The table 12 shows the semi-qualitative data quality rating (DQR) (based on expert judgement) of the most impacting processes in an early phase of a PEF study of intermediate packaging paper. The data set *DE: Potato starch, at plant* was deemed to be of too low quality (DQR = 2.67), as it scored poor (4) on *Parameter uncertainty* and *Methodological Appropriateness and Consistency* to be used for comparative assertion. Therefore, another dataset was searched for that was compliant with the goal and scope of the study.

It is also recommended that the practitioner is aware of the methodological choices used to build the datasets (may be evaluated under Methodological Appropriateness and Consistency), such as the definition of the system boundaries between nature and technosphere and the time-horizons of the emissions (short-term/long term)

System boundaries between nature and technosphere are especially relevant for food and agricultural product (and products made thereof) as for these products the technosphere is closely interlinked with nature (Schau et al 2008). This includes modelling of pesticides and fertilizers (De Schryver and Galatola 2016).

The time-horizons of the emissions are essential, especially for long lived product and for end-of-life processes (e.g. landfilling, waste handling of residues from mining in raw material extraction) (Guo and Murphy, 2012). Also how radioactive waste is modelled is essential to assess with a time horizon in mind. Often, a distinction is made between long term and short term emission. Where the boundary is set between those (100 years, 300 years, 50 000 years etc.) can therefore be important and should be taken into account when assessing the data quality for methodological appropriateness and consistency (see chapter 4.3.3).

Database providers publish their modelling principle (see e.g. Baitz et al, 2016), which is an essential source for evaluating and assessing the data quality (of background data) in addition to the metadata of the LCI data itself.

*Table 11 Data quality rating in an example of intermediate paper products (EC 2013, Schau and Davidsson 2015).*

[7] In the EF Guide the term "representativeness" is used instead of "coverage" used in ISO 14044 for the geographical and time-related data quality requirement. The term "parameter uncertainty" is used instead of "precision" used in ISO14044 and the term "methodological appropriateness and consistency" is used in the EF Guide instead of "consistency" used in ISO14044.

| | | | Data quality elements | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | | Representativeness | | | | | |
| Quality level | Quality rating | Definition | Technological | Geographical | Time-related | Completeness | Methodological Appropriateness and Consistency | Parameter uncertainty |
| Very good | 1 | Meets the criterion to a very high degree, without need for improvement. | E.g. Process is same. For electricity from grid, average technology as country- specific consumption mix. | Country specific data | ≤ 3 years old data | Very good completeness (≥ 90 %) | Full compliance with all requirements of the PEF guide | Very low uncertainty (≤ 7 %) |
| Good | 2 | Meets the criterion to a high degree, with little significant need for improvement. | E.g. average technology as country- specific consumption mix. | Central Europe, North Europe, representative EU 27 mix, | 3-5 years old data | Good completeness (80 % to 90 %) | Attributional Process based approach AND following three method requirements of the PEF guide met: (1) Dealing with multi-functionality; (2) End of life modelling; (3) System boundary. | Low uncertainty (7 % to 10 %) |
| Fair | 3 | Meets the criterion to an acceptable degree, but merits improvement. | E.g. average technology as country- specific production mix or average technology as average EU consumption mix. | EU-27 countries, other European country | 5-10 years old data | Fair completeness (70 % to 80 %) | Attribution Process based approach AND two of the following three method requirements of the PEF guide met: (1) Dealing with multi-functionality; (2) End of life modelling; (3) System boundary. | Fair uncertainty (10 % to 15 %) |
| Poor | 4 | Does not meet the criterion to a sufficient degree, but rather requires improvement. | E.g. average technology as country- specific consumption mix of a group of similar products | Middle east, North-America, Japan etc. | 10-15 years old data | Poor completeness (50 % to 70 %) | Attributional Process based approach AND one of the following three method requirements of the PEF guide met: (1) Dealing with multi-functionality; (2) End of life modelling; (3) System boundary. | High uncertainty (15 % to 25 %) |
| Very poor | 5 | Does not meet the criterion. Substantial improvement is necessary. | E.g. other process or unknown, not available (n.a.) | Global data or unknown | ≥ 15 years old data | Very poor or unknown completeness (< 50 %) | Attributional Process based approach BUT: None of the following three method requirements of the PEF guide met: (1) Dealing with multi-functionality; (2) End of life modelling; (3) System boundary. | Very high uncertainty (>25 %) |

*Table 12 Data quality rating in an early phase of a PEF study of intermediate packaging paper*

| | Representativeness | | | | | | |
|---|---|---|---|---|---|---|---|
| **Data set** | **Technological** | **Geographical** | **Time-related** | **Completeness** | **Parameter uncertainty** | **Methodological Appropriateness and Consistency** | **Resulting Data Quality Rating (DQR)** |
| CH: building, hall, steel construction | 1 | 3 | 3 | 1 | 3 | 4 | 2.50 |
| CH: disposal, sludge from pulp and paper production, 25% water, to sanitary landfill | 1 | 3 | 3 | 1 | 3 | 4 | 2.50 |
| CH: disposal, steel, 0% water, to municipal incineration | 1 | 3 | 3 | 1 | 3 | 4 | 2.50 |
| DE: lignite briquettes, at plant | 1 | 3 | 3 | 1 | 3 | 4 | 2.50 |
| DE: maize starch, at plant | 1 | 3 | 3 | 1 | 3 | 4 | 2.50 |
| DE: potato starch, at plant | 1 | 3 | 3 | 1 | 4 | 4 | 2.67 |
| RER: AKD sizer, in paper production, at plant | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: aluminium sulphate, powder, at plant | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: building, multi-storey | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: electricity, medium voltage, production RER, at grid | 1 | 2 | 2 | 1 | 3 | 3 | 2.00 |
| RER: facilities, chemical production | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: heavy fuel oil, at regional storage | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: heavy fuel oil, burned in industrial furnace 1MW, non-modulating <u-so> | 1 | 2 | 3 | 4 | 3 | 4 | 2.83 |
| RER: industrial wood, hardwood, under bark, u=80%, at forest road | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: industrial wood, softwood, under bark, u=140%, at forest road | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: natural gas, high pressure, at consumer | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: paper mill, integrated <u-so> | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: PEFEmissions-RF <u-so> | 1 | 2 | 3 | 3 | 3 | 3 | 2.50 |
| RER: PEFEmissions-VF v.03 <u-so> | 1 | 2 | 2 | 2 | 4 | 3 | 2.33 |
| RER: sodium chlorate, powder, at plant | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: sodium dithionite, anhydrous, at plant | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: sodium hydroxide, 50% in H2O, production mix, at plant | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: steel, converter, chromium steel 18/8, at plant | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: steel, electric, chromium steel 18/8, at plant | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: sulphate pulp, ECF bleached, at plant | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: sulphate pulp, unbleached, at plant | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: sulphuric acid, liquid, at plant | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: transport, freight, rail | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: transport, lorry >16t, fleet average | 1 | 2 | 3 | 1 | 3 | 4 | 2.33 |
| RER: Water Withdrawals-VF <u-so> | 1 | 1 | 2 | 1 | 3 | 3 | 1.83 |

## 4.3 Further completeness and consistency checks

In this section some further checks are presented, referring both to completeness and consistency checks.

### 4.3.1 Anomaly assessment

An anomaly assessment can be done where, based on previous experience, unusual or surprising deviations from expected or normal results are observed and examined for relevance (Skone, 2000).

As a practical example, it can be considered the case of LCA studies of mining activities. When the production of e.g. copper concentrates was assessed adopting an impact assessment method such as ILCD and performing the hotspot analysis as described in chapter 3 of this report, it was found that the identification of most relevant impact categories did not include Resource Depletion, fossil, mineral. In addition, the most relevant processes and life cycle stages contributing to this impact category were not related to the ones involved in the copper extraction. In the case under analysis, it was found that elementary flows related to extraction of resources from nature were missing from the inventory, because the dataset used in the analysis did not include in its goal and scope the assessment of Resource Depletion, fossil, mineral.

A learning was that it is true that it is important to evaluate what is most relevant (e.g. impact categories, life cycle stages, processes, elementary flows), however it is also important to check if what is found as being non-relevant can be actually considered as such. Hence, it is important to evaluate if the LCI modelled is complete with regard to the impact categories that the assessor wants to evaluate.

### 4.3.2 Cut-off

ISO 14044 identifies cut-off as the "specification of the amount of material or energy flow or the level of environmental significance associated with unit processes or product system to be excluded from a study". ISO 14044 also requires that the effect on the outcome of the study of the cut-off criteria selected shall be assessed and described in the final report.

Cut-off criteria can be based on:

- Mass: e.g. <5% cumulative mass input
- Energy: e.g. <5% cumulative energy input
- Environmental significance: e.g. <5% of the results of the selected impact categories

It is important to note that the cut-off criteria need to be selected based on the goal and scope of the study. Therefore, if the study is intended to be used for comparisons and comparative assertions, a sensitivity check using all the above three criteria is a mandatory requirement.

### 4.3.3 Long term emissions

Long term emissions are those emissions considered to happen beyond 100 years, up to thousands of years (e.g. 60000).

Not all databases include long-term emissions within the system boundaries. In some cases, it can be observed that LCA results, after characterization, normalization and weighting are by far dominated by toxicity-related impact categories. When this is the case, it is recommended to check if the datasets used include long term emissions, which in terms of absolute amounts are well superior compared to emissions occurring within 100 years. It has also to be noted that the models to estimate long term emissions are affected by a high uncertainty which has to be taken into account in the LCA study, to avoid biased conclusions and wrong recommendations.

When dealing with long term emissions, it is important however not to overlook their potential impact, therefore it is recommended to:

- Assess a baseline scenario, excluding long-term emissions
- Perform a sensitivity analysis, including long-term emissions in the LCI.
- Include an analysis of the uncertainty associated to the inclusion of long-term emissions.

# 5 Interpretation on Life Cycle Impact Assessment (LCIA) level

Life Cycle Impact Assessment is the phase in an LCA where the inputs and outputs of elementary flows that have been collected and reported in the inventory are translated into impact indicator results. LCIA has two mandatory steps, which are the classification and the characterization of LCI results and two optional steps: normalisation and weighting.

Many elements can be subject to interpretation in the LCIA phase. In this report the focus will be on:

- the relevance of applying different characterization models,
- the relevance of applying different normalisation sets,
- the relevance of applying different weighting sets,
- the identification of the most sensitive elements in determining the final results, conclusions and recommendations,
- the coverage of characterization factors compared to the inventoried elementary flows

Those points will be illustrated in the following section through illustrative case studies.

First, a case study dealing with sensitivity analysis of using different characterization models for toxicity-related and resource-related impact categories will be discussed.

Second, a case study dealing with using different impact assessment methods. This case study will also be used to go through sensitivity analysis in relation to normalisation sets and weighting sets.

Third, a case study mapping the coverage of characterization factors of toxicity-related categories compared to the inventoried elementary flows will be shown.

## 5.1 Characterization: sensitivity analysis

The use of different characterization models to assess a certain impact category, may play a relevant role in the calculation of the final results and the identification of hotspots, most relevant life cycle stages, processes and elementary flows.

### 5.1.1 Case study 1 – WEEE management: Toxicity-related and Resource-related impact categories affecting interpretation of results

The first case study is based on a work by Rigamonti et al. (2017); the authors selected a case study in which LCA has been applied to assess a waste electrical and electronic equipment (WEEE) management system (Biganzoli et al., 2015). In that study, the assessment was carried out to quantify the mass balance of the WEEE management system in Lombardy Region in the year 2011 and to calculate its environmental benefits and burdens.

Firstly, they tested characterization models and characterization factors (CFs) for toxicity-related impact categories: human toxicity cancer and non-cancer and eco-toxicity. There is an evolving debate on the robustness of these impact categories and specifically for what concern impacts due to metals (see e.g. Pizzol et al., 2011). In fact, some specific features of the metals (e.g. essentiality), as well as elements affecting their fate modelling (e.g. different conditions affecting their bioavailability), are not fully captured by currently available models applied in LCA. Therefore, the authors ran a sensitivity analysis where updated CFs for metals were tested, along the lines of recent literature (Dong et al., 2014) aiming at identifying the different results associated to metals impact, compared to the ILCD method (EC-JRC, 2011), based on the USEtox model (Rosenbaum et al. 2008).

The total impact obtained by applying Dong et al. 2014 CFs resulted about three times higher than the corresponding obtained by the USEtox model. This is mainly due to the Dong et al. 2014 CFs of Copper (Cu) and Zinc (Zn) which are higher than those adopted in the USEtox model when considering the freshwater archetype proposed in the paper of Dong as default when the location of the emission is not known.

Secondly, they tested impact assessment models for resource. Models currently available differ both in the modeling approach, in the perspective adopted for assessing the resources (Dewulf et al., 2015) and, as consequence, in the characterization metric and factors adopted (Mancini et al., 2016). Beyond the potential benefit associated to a mass based approach to recycling (e.g. kg of material recycled), there is indeed the need of understanding to which extent the recycled materials are contributing to the resource depletion impact categories. The authors tested: abiotic depletion potential (ADP) (CML, 2012), which is focusing on potential depletion based on the ratio between resource consumption and availability (either considering ultimate reserves in earth crust, known base reserves, or economically viable reserves); ILCD model (EC-JRC, 2011), which is an extended version of CML2012 reserve based (based on CML algorithm, few other resources have been added described in Sala et. al, 2012); EDIP97 (Hauschild and Wenzel, 1998), which is comparing the resource with the deposits economically exploitable, without accounting for current level of consumption; EPS2000 (Steen, 1999), which assesses the cost (as society's willingness to pay) of substituting a substance by an alternative for future generations affected by current level of depletion; Recipe 2008 (Goedkoop et al., 2009), which is looking at marginal increase of extraction cost per kg of extracted resource, differentiating it by deposit and assuming a discount rate over an indefinite timespan; the Anthropogenic Abiotic Depletion Potential (AADP, Schneider et al., 2011), which is accounting for the potential of resource recycling, assuming urban mining as an additional source of resources. The recent update of the AADP (Schneider et al., 2015) was also considered. It introduces the concepts of "ultimately extractable reserves" represented by the amount available in the upper earth's crust that is ultimately recoverable. A new set of CFs were also tested, developed by Mancini et al. (2016) aiming at addressing the benefit associated to the recovery of Critical Raw Materials (CRMs). The characterization model is based on the use and adaptation to LCA of the supply risk indicators developed by the European Commission (EC, 2014). Three sets of CFs are tested, based on different assumptions: (1) baseline option, the supply risk factors as such – (SR); (2) an exponential function which magnifies the differences between the CRMs - (SR)^6; (3) the ratio between supply risk and production data (SR/world mine production in 2011), which reflects the size of the market, giving more importance to the materials used in small amounts in products and applications, like, e.g., specialty metals, that are often perceived as critical.

The comparison among the different models was first of all made considering the mineral resources present in the inventory of the case study covered by each of them. The ILCD method is the most comprehensive one with 31 mineral resources included in its model for the resource depletion category. On the contrary, the Supply Risk (3) is the one that covers the minor number of substances (i.e. 7).

A contribution analysis was also performed (Figure 3). Silver (Ag) gives the main percentage contribution to the indicator in five models (i.e. CML 2012 versions b and c, ILCD, Supply Risk 3), and EDIP97). Cobalt (Co) is the main contributor in the Supply Risk (2), Copper (Cu) in the ReCiPe 2008, Palladium (Pd) in the EPS 2000 and AADP (2015), and Nickel (Ni) in the AADP (2011). In the CML 2012 (a) Tellurium, Gold (Au) and Silver (Ag) are the most important substances, whereas Aluminium (Al), Copper (Cu) and Nickel (Ni) are the main contributors in the case of the Supply Risk (3).

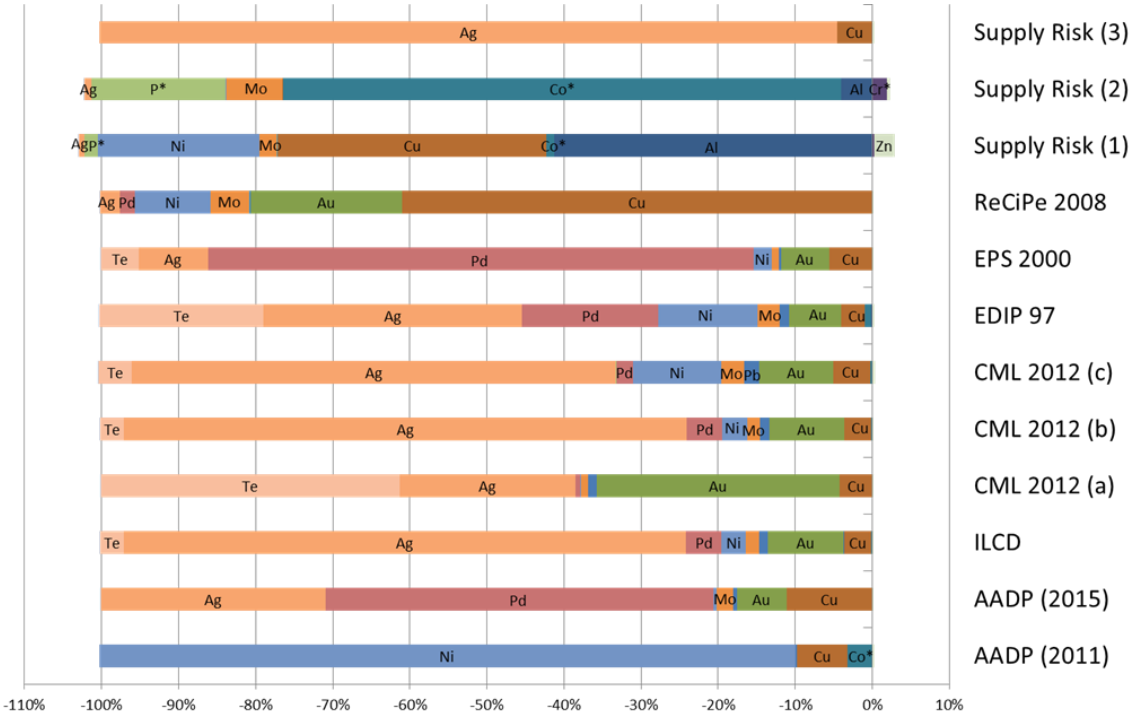


*Figure 3 Contribution analysis for the resource- related indicator calculated adopting several characterization models (Rigamonti et al., 2017)*

### 5.1.2 Case study 2 – Basket of Products Food: sensitivity analysis comparing different impact assessment methods

This case study is based on the application of different life cycle impact assessment methods for the evaluation of the environmental impacts associated to food consumption in Europe. A process-based LCA has been performed for a basket of products that represent the most relevant food product groups, selected by importance in mass and economic value, to depict the average consumption for nutrition of EU citizens in 2010 (Notarnicola et al. 2017b). The product groups in the basket were: pig meat, beef, poultry, milk, cheese, butter, bread, sugar, sunflower oil, olive oil, potatoes, oranges, apples, mineral water, roasted coffee, beer, pre-prepared dishes. For each product group in the basket, an inventory model based on a representative product has been developed. The impact of each representative product was then multiplied by the mass of products in that product group that is consumed in one year by an average EU citizen.

The environmental impact of the average food consumption of a European citizen has been characterized using ILCD method for the life cycle impact assessment (EC-JRC, 2011). The paper by Notarnicola et al. (2017) illustrates the details of the process-based LCA study on the BoP nutrition for EU 27 citizens and presents the results of the characterisation phase. The variability of the results associated to methodological choices has been already preliminary explored (Castellani et al. 2017). The present report builds on those results and elaborates the interpretation phase (including also normalisation and weighting). The questions that these analyses help answering are: i) are the products found to have the highest impact sensitive to changes in LCIA elements (characterisation factors, normalization references or the weighting sets)? and ii) which are the most contributing impact categories?

This case study will also be used in paragraph 5.2, 5.3 and 5.4 as a basis for further insights on the interpretation of an LCA study.

For this specific case, the authors performed a sensitivity analysis on the LCIA characterisation phase, using ILCD compared with other two methods: the CML-IA, v 4.2 (Guinée et al. 2002) and the Ecological footprint (EcF) (Wackernagel et al. 2005). CML-IA was chosen because it is widely used in LCA studies and because it adopts a different

modelling approach compared to ILCD in several impact categories that could be relevant for food chains. For instance, for toxicity-related impact categories CML-IA is based on USES-LCA model (Van Zelm et al. 2009), whereas ILCD is applying USETOX model (Rosenbaum et al. 2008); for eutrophication, CML-IA is using RAIN-LCA (Huijbregts, 1999) whereas ILCD is using different models for terrestrial and aquatic eutrophication (respectively accumulated exceedance by Seppälä et al. 2006, and EUtrend model as applied by Struijis et al. 2009). The EF was chosen because it assesses impacts in terms of direct and indirect land use and has been considered a method that may complement LCA (Lee et al. 2014) although based on an accounting framework that has been questioned (Giampietro and Saltelli 2014). The version of EcF method implemented in the study is the one already adapted to LCA, and provided within the SimaPro 8.0.5 software package (Prè Consultants, 2015).

The three methods are mostly convergent in the identification of the product groups to be considered hotspots within the BoP nutrition, i.e. within the food consumption of the European citizens. ILCD and CML-IA identify a wider range of hotspots compared to EcF, mainly because they cover a wider range of impact categories (e.g. bread and potatoes do not emerge as hotspot for EcF, because their main impacts are in impact categories not covered by the EcF method).

The product groups that emerge as hotspots in all the methods, even if with slightly different level of contribution, are meat and dairy products and beer (as representative product for alcoholic drinks).

The impact categories for which beer emerge as hotspot are slightly different among the three methods: ozone depletion, ionizing radiation and resource depletion in ILCD and CML-IA, photochemical ozone formation in ILCD, human toxicity and marine aquatic ecotoxicity in CML-IA and nuclear energy in EcF. The most contributing processes for all the impact categories listed before is energy (electricity and heat), as already discussed for ILCD. In the case of CML-IA method, toxicity-related impacts are mainly due to direct emissions to air coming from the production process of the packaging glass.

### 5.1.3 Conclusions and recommendations

These results reveal that the relative importance of one substance compared to another changes with the model. This is because the models differ in the substances that they include and in the CFs of each substance. It is recommended to run a sensitivity check with different characterization models to evaluate the robustness of the conclusions drawn. It is also recommended to evaluate the theory underlying the characterization models chosen to evaluate a certain impact category, as this helps understanding the results obtained.

## 5.2 Characterization: mapped flows and unmapped flows

This section shows two different levels of checks: the first one focuses on a comparison between two different life cycle impact assessment methods (i.e. ILCD vs CML-IA) based on Castellani et al. (2017), the second one, focuses on what are the implications of uncharacterized flows for a specific impact category (freshwater ecotoxicity), based on the case study presented in Notarnicola et al. (2017b).

### 5.2.1 Comparison between two different methods

A check of the inventoried elementary flows that are not characterized at the LCIA phase should always be done to support interpretation of results. Indeed, the comprehensive translation of the inventory into potential environmental impacts, may not occur if some of the elementary flows are not covered by the chosen characterization models. Table 13 reports an example of the total amount of elementary flows that are available in an inventory but they are not mapped, and characterized, by the methods considered (ILCD and CML-IA), as reported in Castellani et al. (2017). The authors checked the list of uncharacterized flows for the inventory of the case study under evaluation and

highlighted the most relevant ones, either in term of mass or of known environmental concern. The information obtained was then used in support to interpretation. For instance, in case the two methods used for characterisation and normalisation give very different results for one impact category, the information about uncharacterized flows can help to check if this difference is due to a different characterisation of impacts (i.e. to a different way of considering the environmental relevance of some elementary flows) or if it is affected by the fact that some flows (e.g. some emissions) are not taken into account by one of the methods.

This may occur also due to improper nomenclature of the elementary flows, which may cause a mismatch between the elementary flows listed in the inventory and the elementary flows to which a characterization factor is available: for example, the inventory may report $CaCO_3$, while the characterization model may provide a CF for calcite. This mismatch would prevent the characterization of $CaCO_3$.

*Table 13 Example of check of inventoried elementary flows that are not characterized by two impact assessment methods (based on Castellani et al. 2017)*

| **Flows** | **Total elementary flows not characterized** | | **Typologies of flows** | **Details on the elementary flows not characterized** | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | **ILCD** | **CML-IA** | | **ILCD** | | | **CML-IA** | | |
| | | | | **Nr flows** | **Amount** | **Unit** | **Nr flows** | **Amount** | **Unit** |
| raw materials | 97 | 510 | metals and minerals | 47 | 79.5 | kg | 47 | 86.3 | kg |
| | | | water flows | 25 | 1.26 | $m^3$ | 300 | 382.5 | $m^3$ |
| | | | Energy | 0 | 0 | MJ | 12 | 1896.4 | MJ |
| | | | land use | 11 | 0.01 | $m^2$ (or $m^2a$) | 117 | 1289.8 | $m^2$ (or $m^2a$) |
| | | | *others* | *14* | n.a. | *mixed* | 34 | n.a. | *mixed* |
| emissions to soil | 83 | 227 | pesticides to soil | 45 | 0.003 | kg | 173 | 0.08 | kg |
| | | | *others* | 38 | 0.8 | kg | 54 | 4.1 | kg |
| emissions to water | 167 | 310 | radioactive substances | 42 | 1.19E+03 | Bq | 61 | 3.85E+06 | Bq |
| | | | metals and inorganics compounds | 34 | 2.8 | kg | 36 | 2.8 | kg |
| | | | *others* | 91 | n.a. | *mixed* | 213 | n.a. | *mixed* |
| emissions to air | 179 | 286 | radioactive substances | 49 | 8.99E+04 | Bq | 73 | 9.73E+07 | Bq |
| | | | metal and inorganics compounds | 65 | 0.07 | kg | 56 | 0.044 | kg |
| | | | pesticides | 7 | 5.80E-08 | kg | 32 | 9.3 | kg |
| | | | *others* | 70 | n.a. | *mixed* | 125 | n.a. | *mixed* |
| waste flows | 17 | 17 | waste | 17 | 37.84 | kg | 17 | 37.84 | kg |
| **Overall number of elementary flows not characterized** | **543** | **1350** | | | | | | | |

In order to assess which elementary flows available in the inventory were not accounted during the characterization, the authors performed an analysis of the elementary flows not characterized when using ILCD and CML-IA methods. In total, the model of the basket includes 1730 elementary flows. The ILCD method is not able to cover 543 of these elementary flow as emission into air, water, soil or resource used, whereas when CML-IA is used to characterize the inventory, 1350 elementary flows out of 1730 are not

characterized. In Table 1 the list of main typologies of flows is reported. Many pesticides, organic chemicals and inorganics are not characterized by the two methods. Several ionizing substances are not characterized as well as elementary flows related to group of substances (such as AOX, PAH, BTEX, VOC, BOD and COD). Missing the CFs means that it is not possible to estimate the extent to which e.g., the toxicity-related and ionizing radiation-related impact categories are underestimated.

Apart from the total number of flows covered, the main difference between the two LCIA methods is the ability to assign a characterization factor (CF) to the pesticides included in the inventory. ILCD has no CFs for 45 pesticides' emission to soil (plus 4 pesticides' emissions to air), whereas CML-IA for 173 (plus 32 pesticides' emissions to air). This may be the reason why human toxicity impacts appear as relevant in the results of ILCD characterization (and related normalization and weighting, as discussed below) and they appear as less relevant in the results of CML-IA characterization and related normalization and weighting.

### 5.2.2 Analysis of uncharacterized flows within an impact category

Based on the study of Castellani et al. (2017), fourteen impact categories (IC) recommended in PEF are calculated. The number of elementary flows (ElFls) by ICs varies from few flows (eutrophication, acidification, etc..) to thousands (eco and human toxicity) (see table 14).
For most of the ICs, the contributing elementary flows reported in an inventory are well identified and their inclusion in the calculation of the final score is expected to be done correctly. For climate change at mid-point, 104 elementary flows with their associated CFs are derived from the IPCC 2007 report for a 100-year period.
However, for impact categories like toxicity, the number of elementary flows contributing to the impact can be very important making possible the omission of some EFs at the inventory level. Additionally, a high number of flows reported in the inventory have no CFs. If the number of un-characterized flows is too high, this can lead to a non-robust interpretation of the LCIA results.

Table 12 shows the number of elementary flows with associated a CF (i.e. the single chemical) and the amount of elementary flows with a CF per different compartment (e.g. the single chemical in each compartment and sub-compartment)

*Table 14 List of impact categories listed in ILCD with total number of elementary flows contributing to each IC.*

| **Impact Category** | **ElFl with CFs (all emission compartments)** | **Unique ElFls with CFs (single chemical / flows)** |
|---|---|---|
| Climate change | 104 | 102 |
| Ozone Depletion | 23 | 23 |
| Human toxicity non cancer | 3434 | 442<br>(USEtox 419 org+18 metals) |
| Human toxicity cancer | 4734 | 607<br>(USEtox 596 org +8 metals) |
| Particulate matter | 46 | 24 |
| Ionizing radiation Human Health | 50 | 29 |
| Photochemical ozone formation | 133 | 133 |
| Acidification | 7 | 7 |
| Terrestrial eutrophication | 7 | 7 |
| Freshwater eutrophication | 10 | 5 |
| Freshwater ecotoxicity | 15636 | 2526<br>(USEtox 2498 org +21 metals) |
| Land use | 206 | 206 |
| Water source depletion | 1180 | 1180 |
| Mineral, fossil use & Resource depletion | 181 | 181 |
| *Marine eutrophication* | *14* | *11* |
| *Additional IC not used in PEF* | | |
| *Ionizing radiation Ecosystem (interim)* | *62* | *15* |

The potential impact of un-mapped (or uncharacterized) flows is illustrated via a case study focusing the freshwater aquatic toxicity impact category using a recent modelling of 17 food products (Notarnicola, 2017b).

For a high proportion of the total mass of un-characterized ElFls, the impact on the final IC scores is null. As a matter of fact, 99.98% of the 1887 kg ElFls emitted to the air are made of: air, oxygen and hydrogen. Only 0.33 kg (but still 199 flows) are actual made of other type of chemicals: 80% of those 0.33 kg are nitrogen type of chemicals, 7.4% hydrogen chloride, 1.2% sulphate, etc. (table 15). However, regarding the ElFls to soil and water, the mass and number of un-characterized flow could have a significant impact on the final score.

*Table 15 Type of chemicals (mass and number of EFs) not characterized per emission compartment (air, oxygen and hydrogen have been excluded).*

| Compartments | Group of chemicals | Mass (g) | Number of ElFl |
|---|---|---|---|
| Air | **Total air** | **330** | **122** |
| | N-related | 238.662 | 2 |
| | inorganic/salt | 74.637 | 44 |
| | organic | 5.693 | 36 |
| | S - related | 3.896 | 1 |
| | metal/mineral | 3.793 | 17 |
| | Alkenes | 1.530 | 2 |
| | Benzene, Toluene, Ethylbenzene, and Xylene), unspecified ratio | 0.488 | 1 |
| | ozone | 0.314 | 1 |
| | P-related | 0.304 | 4 |
| | ionising | 0.261 | 4 |
| | Aldehydes | 0.213 | 1 |
| | Terpenes | 0.102 | 1 |
| | PAH | 0.034 | 1 |
| | Pesticide | 0.000 | 6 |
| | Acid | 0.000 | 1 |
| Soil | **Total soil** | **800** | **68** |
| | metal/mineral | 328.315 | 15 |
| | inorganic | 320.234 | 4 |
| | oils | 93.496 | 3 |
| | organic | 48.702 | 2 |
| | Pesticide | 5.873 | 39 |
| | S - related | 3.777 | 2 |
| | N-related | 0.000 | 2 |
| | PAH | 0.000 | 1 |
| Water | **Total water** | **13871** | **112** |
| | inorganic/salt | 7549.785 | 22 |
| | TOC water | 2840.410 | 4 |
| | metal/mineral | 1998.366 | 13 |
| | Suspended solid | 1257.655 | 6 |
| | inorganic | 108.851 | 6 |
| | organic | 107.546 | 37 |
| | AOX | 2.390 | 1 |
| | Acid | 2.032 | 2 |
| | S - related | 2.008 | 5 |
| | VOC | 0.643 | 1 |

| | | | |
|---|---|---|---|
| | N-related | 0.511 | 3 |
| | Propylene | 0.244 | 1 |
| | oils | 0.079 | 2 |
| | PAH | 0.055 | 2 |
| | Sodium Hydroxide | 0.054 | 1 |
| | Salt, unspecified | 0.015 | 1 |
| | Hypochlorite | 0.013 | 1 |
| | ionising | 0.002 | 2 |
| | VOC | 0.001 | 1 |
| | P- related | 0.000 | 1 |

Looking at the results of the freshwater aquatic toxicity impact categories for all 17 food products, 90% of the impact category is driven by 10 out of 734 elementary flows that represents only 1,4% of the mass of chemicals entering the aquatic environment (table 16). Furthermore, it can be seen that:

- 57% of the toxicity score are due to 3 metals representing 0,1 kg of the mass. However, there are still more than 2 kg of un-characterized metals/minerals in the inventory list (see table 15).
- 33% of the impact are due 10 pesticides (35 g), although there are still 39 un-characterized pesticides in the inventory list (representing 5 g) (see table 15).

*Table 16 Most relevant elementary flows, contributing cumulatively up to 90% to the freshwater ecotoxicity impact category with USEtox.*

| Substance | Chemical class | Emission Compartment | Sub-compartment | Total impact (CTUe) | % impact versus Total | % Cumulated impact | Mass (kg) |
|---|---|---|---|---|---|---|---|
| Zinc | metal | Soil | agricultural | 931 | 21% | 21% | 0.0441 |
| Copper | metal | Soil | agricultural | 902 | 21% | 42% | 0.0429 |
| Chlorpyrifos | pesticide | Soil | agricultural | 833 | 19% | 61% | 0.0079 |
| Copper | metal | Soil | | 351 | 8% | 69% | 0.0120 |
| Zinc | metal | Water | | 139 | 3% | 72% | 0.0036 |
| Folpet | pesticide | Soil | agricultural | 134 | 3% | 75% | 0.0004 |
| Chlorothalonil | pesticide | Soil | agricultural | 111 | 3% | 78% | 0.0019 |
| Chromium | metal | Water | | 105 | 2% | 80% | 0.0020 |
| Cyfluthrin | pesticide | Soil | agricultural | 85 | 2% | 82% | 0.0001 |
| Isoproturon | pesticide | Soil | agricultural | 81 | 2% | 84% | 0.0086 |
| Cypermethrin | pesticide | Soil | agricultural | 70 | 2% | 85% | 0.0010 |
| Chromium | metal | Soil | agricultural | 51 | 1% | 87% | 0.0019 |
| Prochloraz | pesticide | Soil | agricultural | 45 | 1% | 88% | 0.0024 |
| Captan | pesticide | Soil | agricultural | 39 | 1% | 88% | 0.0039 |
| Alachlor | pesticide | Soil | agricultural | 38 | 1% | 89% | 0.0041 |
| Metolachlor | pesticide | Soil | agricultural | 30 | 1% | 90% | 0.0050 |

Among the non-characterized elementary flows, in relation to freshwater ecotoxicity, it can be observed:

- it is not known to which extent some of the chemicals may contribute to the IC;
- a number unspecified chemicals such as 'Total organic matter' (TOC, BOD, COD…), PAH, VOC that could contribute significantly to the overall score. Since the quantity of those ElFl can change significantly between products, a default CF could be proposed for those flow
- some chemicals are known to be contributing to toxicity like pesticides and metals, still there are significant proportions that are not characterized.

Although it is not possible to conclude if those un-characterized flows would have changed the total score significantly, based on the mass and number of known-contributing class of chemicals (metals and pesticides) it is highly recommended that a quantification of the uncharacterized chemicals is reported for the freshwater toxicity impact category before drawing final conclusions.

### 5.2.3 Conclusions and recommendations

Results of the LCIA phase can be highly affected by the characterization model chosen, as shown in chapter 5.1 Furthermore, the evaluation of uncharacterized elementary flows is of primary importance to assess the completeness of the LCIA coverage of the identified LCI flows. Indeed, a high number of elementary flows inventoried might be non-characterized by the chosen models, even if their relevance in the assessed impact categories could be of primary importance. Therefore, it is highly recommended that a quantification of the uncharacterized elementary flows is reported before any interpretation or conclusion is performed on the superiority of one product versus another.

## 5.3 Normalization: sensitivity analysis

As for the characterisation phase, also the choice of the normalisation reference set can highly influence results. Castellani et al (2017) have done a sensitivity analysis based on the case study of a basket of product on food (see paragraph 5.1.2 for further details).

The authors report that as an optional step, normalisation and weighting could be applied to characterized LCA results to support the identification of the most important impact categories. Given that several sets of normalisation and weighting are available, the authors tested some of them in addition to the ILCD baseline, with the aim of assessing the sensitivity of the results to the choice of particular geographical and temporal reference systems (i.e. normalization using EU or global scales) as well as perspectives (i.e. weighting). This was investigated by selecting 3 sets of ILCD-compliant normalization references (Benini et al. 2014, Benini et al., 2015, Laurent et al., 2013) and 2 sets of CML-IA compliant normalization references (EU 25+3, year 2000 and World, year 2000 - Guinée et al., 2002), -reported in Table 17. In this section, the findings of the authors regarding normalisation will be discussed, while the findings on weighting will be discussed in section 5.4.

*Table 17 Summary of normalization sets used in the sensitivity analysis in Castellani et al. (2016)*

| ILCD Impact Category | Unit | EC-JRC EU27 (2010), per person[a] | EC-JRC Global (2010 or 2013), per person[b] | PROSUITE Global (2010 or 2000), per person[c] |
|---|---|---|---|---|
| Climate change | kg $CO_2$ eq. | 9.22E+03 | 7.07E+03 | 8.10E+03 |
| Ozone depletion | kg CFC-11 eq. | 2.16E-02 | 1.22E-02 | 4.14E-02 |
| Human toxicity, cancer effects | CTUh | 3.69E-05 | 1.24E-05 | 5.42E-05 |
| Human toxicity, non-cancer effects | CTUh | 5.33E-04 | 1.55E-04 | 1.10E-03 |
| Particulate matter/Respiratory inorganics | kg PM2.5 eq. | 3.80E+00 | 5.07E+00 | 2.76E+00 |
| Ionizing radiation, human health | kBq $U^{235}$ eq. (to air) | 1.13E+03 | 2.41E+02 | 1.33E+03 |
| Photochemical ozone formation, human health | kg NMVOC eq. | 3.17E+01 | 4.53E+01 | 5.67E+01 |
| Acidification | mol H+ eq. | 4.73E+01 | 5.61E+01 | 4.96E+01 |
| Eutrophication terrestrial | mol N eq. | 1.76E+02 | 1.64E+02 | 1.15E+02 |
| Eutrophication freshwater | kg P eq. | 1.48E+00 | 6.54E+00 | 6.20E-01 |
| Eutrophication marine | kg N eq. | 1.69E+01 | 3.04E+01 | 9.38E+00 |
| Land use | kg C deficit | 7.48E+04 | 5.20E+06 | 2.36E+05 |
| Ecotoxicity freshwater | CTUe | 8.74E+03 | 3.74E+03 | 6.65E+02 |
| Resource depletion water | $m^3$ water eq. | 8.14E+01 | 6.89E+01 | 2.97E+01 |
| Resource depletion, mineral, fossils and renewables | kg Sb eq. | 1.01E-01 | 1.93E-01 | 3.13E-01 |

An additional analysis was performed to verify which one between normalization and weighting is the most sensitive step in the identification of the most relevant impact categories. The same weighting sets described before were applied on the results of normalization obtained using the two ILCD-compliant global normalization reference sets tested before, namely EC-JRC Global (Benini et al., 2015) and PROSUITE Global (Laurent et al., 2013).

The contribution of product groups (Table 18) and of impact categories (Table 19) to the total normalized impact of the BoP nutrition per citizen were analyzed.
Table 18 presents a summary of the sensitivity analysis on product group contribution at the normalization stage, comparing the product group hotspots according to ILCD and CML-IA methods. Values are expressed as percentage of the normalized value per citizen for that product group with respect to the total normalized impact per citizen (total impact of nutrition in one year for an average EU-27 citizen). Equal weighting among impact categories is assumed. Results at the normalization stage almost confirm the hotspots identified at the characterization stage: meat and dairy products are again hotspots for all the methods, whereas other products that emerged before are considered less contributing at the normalization stage. This means that those products generate impacts in impact categories that are considered less relevant compared to the others, according to the normalization scheme adopted. For instance, beer appeared to be a hotspot for both ILCD and CML-IA at the characterization stage, whereas only CML-IA method identifies it as a hotspot at the normalization stage. This happens because the CML-IA normalization references (EU 25+3 and World) for marine aquatic toxicity -one of the impact categories for which beer is a hotspot - are relatively low compared to the ones in other LCIA methods.
Table 18 presents a summary of the results of the sensitivity analysis at the normalization stage, comparing the relevance of impact categories according to ILCD and CML-IA methods. Results refer to the entire BoP. For each impact category of the two methods, values are expressed as percentage of the normalized value per citizen for that impact category with respect to the total normalized impact per citizen.

*Table 18 Results of sensitivity analysis on normalization sets at the European and global scale, for ILCD- and CML-IA methods (Castellani et al., 2016). The percentages represent the contribution of each product group to the total impacts of the BoP per citizen (reference flow). Equal weighting of impact categories is applied. The second column reports the mass of each product group in the basket. A color scale is applied to the results in each column (i.e. product group), from green (lowest contribution), to red (highest contribution).*

| | Quantity in the basket (kg/year) | Share of total BoP mass | EC-JRC EU27 (2010) | EC-JRC Global (2010 or 2013) | PROSUITE Global (2010 or 2000) | CML EU 25+3 (2000) | CML World (2000) |
|---|---|---|---|---|---|---|---|
| Mineral water | 105.0 | 19.4% | 1% | 1% | 1% | 3% | 3% |
| Beer | 69.8 | 12.9% | 4% | 4% | 7% | 16% | 21% |
| Coffee | 4.7 | 0.6% | 2% | 1% | 5% | 5% | 6% |
| Apple | 16.1 | 3.0% | 0% | 0% | 1% | 1% | 1% |
| Oranges | 17.4 | 3.2% | 1% | 1% | 2% | 1% | 2% |
| Potatoes | 69.1 | 13.0% | 3% | 3% | 3% | 3% | 4% |
| Bread | 39.3 | 7.3% | 3% | 3% | 3% | 4% | 5% |
| Olive oil | 5.3 | 1.0% | 1% | 0% | 1% | 2% | 3% |
| Sunflower oil | 5.3 | 1.0% | 4% | 4% | 5% | 3% | 3% |
| Sugar from beet | 29.8 | 5.5% | 3% | 3% | 3% | 2% | 1% |
| Milk | 80.1 | 14.8% | 9% | 9% | 8% | 7% | 6% |
| Cheese | 15.0 | 2.8% | 15% | 16% | 13% | 10% | 9% |
| Butter | 3.6 | 0.7% | 7% | 8% | 6% | 3% | 2% |
| Meat - beef | 13.7 | 2.5% | 18% | 17% | 16% | 11% | 10% |
| Meat - pork | 41.0 | 7.6% | 20% | 20% | 19% | 18% | 13% |
| Meat - poultry | 22.0 | 4.2% | 8% | 8% | 8% | 10% | 7% |
| Pre-prepared meal | 2.9 | 0.5% | 1% | 1% | 1% | 2% | 2% |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | | 100% | 100% | 100% | 100% | 100% | 100% |

*Table 19 Results of sensitivity analysis on normalization sets at the European and global scale, for ILCD and CML-IA methods (Castellani et al., 2016). The percentages reflect the relative relevance of impact categories compared to the total impacts of the BoP per citizen (reference flow). A colour scale is applied to the results in each column (i.e. impact category), from green (lowest contribution), to red (highest contribution).*

| | **EC-JRC EU27 (2010)** | **EC-JRC Global (2010 or 2013)** | **PROSUITE Global (2010 or 2000)** | | **CML EU 25+3 (2000)** | **CML World (2000)** |
|---|---|---|---|---|---|---|
| Climate change | 1.9% | 1.1% | 1.3% | Global warming (GWP100a) | 7.8% | 10.2% |
| Ozone depletion | 0.0% | 0.0% | 0.0% | Ozone layer depletion | 0.2% | 0.0% |
| Human toxicity, cancer effects | 6.2% | 8.3% | 2.5% | Human toxicity | 5.9% | 12.0% |
| Human toxicity, non-cancer effects | 44.6% | 69.2% | 12.7% | | | |
| Particulate matter | 2.6% | 0.9% | 2.1% | - | | |
| Ionizing radiation HH | 0.5% | 1.1% | 0.3% | - | | |
| Photochemical ozone formation | 1.0% | 0.3% | 0.3% | Photochemical ozone formation | 4.6% | 2.1% |
| Acidification | 7.3% | 2.86% | 4.1% | Acidification | 27.2% | 20.1% |
| Terrestrial eutrophication | 8.3% | 4.0% | 7.5% | Eutrophication | 13.3% | 16.3% |
| Freshwater eutrophication | 3.4% | 0.4% | 4.8% | | | |
| Marine eutrophication | 8.2% | 2.0% | 8.7% | | | |
| Freshwater ecotoxicity | 5.8% | 6.1% | 45.0% | Freshwater aquatic ecotoxicity. | 34.0% | 31.4% |
| - | - | - | - | Marine aquatic ecotoxicity | | |
| - | - | - | - | Terrestrial ecotoxicity | | |
| Land use | 2.3% | 0.0% | 0.4% | - | | |
| Water resource depletion | 6.2% | 3.3% | 10.0% | - | | |
| Mineral, fossil & ren resource depletion | 1.8% | 0.4% | 0.4% | Abiotic depletion | 0.2% | 1.1% |
| | | | | Abiotic depletion (fossil fuels) | 6.8% | 6.7% |
| | 100% | 100% | 100% | | 100% | 100% |

### 5.3.1 Conclusions and recommendations

Different normalisation sets may produce different results in terms of identification of most relevant impact categories, life cycle stages and processes. Different reference systems refer to different situation which can all be valid, depending e.g. on the geographical scope of the study. It is recommended to evaluate different normalisation tests, which fit the study at hand, to support the conclusions drawn with the baseline scenario or to provide additional insights.

## 5.4 Weighting: sensitivity analysis

Castellani et al. (2016), investigated only for ILCD-compliant normalization sets, in addition to equal weighting, 8 ILCD-compliant weighting sets (reported in Table 20) were tested. The different sets were selected aiming at covering several perspectives on weighting, namely: distance to target for EU policies considering binding and non-binding target at 2020 (Castellani et al., 2017); a previous distance to target set (EDIP)

for Europe for 2005 (Stranddorf et al., 2005); two sets considering planetary boundaries (Tuomisto et al., 2012; Bjørn and Hauschild, 2015[8]); a set which gives relevance to midpoint indicators based on their contribution to impact at the endpoint (Ponsioen and Goedkoop, 2015); a set resulting from the combination of different panel-based approaches (Huppes et al., 2012).

When needed, available weighting sets have been adapted to ILCD impact categories. The majority of these weighting sets build on normalization factors that are specific of a geographic scale. Tuomisto et al. (2012) calculated planetary boundaries at global scale, whereas and Bjørn and Hauschild (2015) did it for both global and European scales. Huppes et al. (2012) adopted a panel-based approach with no specific geographic representativeness, whereas all the other weighting sets used in this work have been originally developed, or adjusted, for applications at the targeting the European scale. The use of EU27-consistent weighting sets in combination with global normalization factors may lead to inconsistencies; therefore, the results have to be interpreted in the light of this inconsistency. In this work the normalized and weighted results are calculated and presented anyway for all combinations so to show the effect of the selection of weighting available for practitioners, rather than focussing on the (in-)consistency of their joint use with normalization factors. The performed analysis quantified the relative contribution by impact category to the overall impacts associated with the BoP, as well as the contribution by each of the products. The effects of the different weighting sets tested on the results of the BoP normalized to EC-JRC EU27 normalization references are presented in Table 21 and Table 22.

*Table 20 Summary of weighting sets used in the sensitivity analysis by Castellani et al. (2016)*

| | Distance to target | | | | | | Damage oriented | Panel-based |
|---|---|---|---|---|---|---|---|---|
| | Policy targets | | | | Planetary boundaries | | Mid-to-endpoint | |
| | Castellani et al. 2016 WFsA | Castellani et al. 2016 WFsB | EDIP 2003 (Stranddorf et al., 2005) | Tuomisto et al. 2012 | Bjørn & Hauschild - European 2015 | Bjørn & Hauschild – Global 2015 | Ponsioen & Goedkoop 2016 | Huppes et al. 2012 |
| **ILCD Impact Category** | **dimensionless (%)** | | | | | | | |
| Climate change | 7.1% | 5.4% | 2% | 10% | 25% | 26% | 44% | 23.2% |
| Ozone depletion | 6.4% | 4.9% | 87% | 8% | 1% | 2% | 0% | 3.6% |
| Human toxicity, cancer effects | 6.9% | 5.2% | 2% | n.a | n.a | n.a | 1% | 6.5% |
| Human toxicity, non-cancer effects | 6.2% | 4.7% | 2% | n.a | n.a | n.a | 4% | 4.1% |
| Particulate matter/Respiratory inorganics | 7.4% | 5.6% | n.a | n.a | n.a | n.a | 8% | 6.6% |
| Ionizing radiation, human health | 6.1% | 4.6% | n.a | n.a | n.a | n.a | 0% | 6.5% |
| Photochemical ozone formation, human health | 7.8% | 5.9% | 2% | n.a | 34% | 48% | 0% | 5.4% |
| Acidification | 7.2% | 5.5% | 2% | 8% | 1% | 1% | 0% | 4.2% |
| Eutrophication terrestrial | 7.0% | 5.3% | 2% | 28% | 1% | 0% | 0% | 2.3% |
| Eutrophication freshwater | 6.2% | 4.7% | 1% | 7% | 9% | 2% | 0% | 2.3% |
| Eutrophication marine | 6.9% | 5.2% | 2% | 28% | 1% | 1% | 0% | 2.3% |
| Land use | 6.4% | 5.3% | n.a | 6% | 25% | 16% | 19% | 10.2% |
| Ecotoxicity freshwater | 6.1% | 5.1% | 0% | n.a | 2% | 0% | 0% | 10.9% |
| Resource depletion water | 6.1% | 29.6% | n.a | 5% | 1% | 4% | 3% | 5.1% |
| Resource depletion, mineral, fossils and renewables | 6.1% | 3.0% | 0% | n.a | n.a | n.a | 19% | 6.9% |

[8] Bjørn and Hauschild provided to the authors of this report a modified version to be used as weighting method in compliance with EU27 ILCD normalization factors. Global weighting factors were calculated directly from global normalization factors published by Bjørn and Hauschild 2015

The contribution of specific products to the overall impacts associated to the BoP nutrition is very similar when using several weighting sets in combination with normalization references EU27 (Table 21). In fact, for all the weighting sets tested pork meat and beef meat are the highest contributor to the impacts (varying between 15% and 22%). According to all sets, cheese is the 3rd largest contributor (14% on average), whereas milk and poultry meat are most frequently identified as 4th and 5th largest contributors (roughly 8% each). On average, these product groups account for roughly 70% of the total impacts. Apples, olive oil and pre-prepared meals are those contributing the less, followed by mineral water and oranges. On average, their cumulative contribution is around 5%.

*Table 21 Product groups contribution. Results of several ILCD-compliant weighting sets in combination with normalization references EU27 (Castellani et al., 2016). A color scale is applied to the results in each column (i.e. product group), from green (lowest contribution), to red (highest contribution).*

| PRODUCT GROUPS CONTRIBUTION | Quantity in the basket (kg/year) | Share of total BoP mass | Equal weighting | Castellani et al. (2016) WFsA | Castellani et al. (2016) WFsB | EDIP2003 [9] | Tuomisto et al. (2012) | Bjorn& Hauschild (2015) – European | Bjorn& Hauschild (2015) – global | Ponsioen and Goodkoep (2016) | Huppes et al. (2012) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Mineral water | 105 | 19.40% | 1% | 1% | 3% | 1% | 1% | 1% | 2% | 1% | 1% |
| Beer 66 cL | 69.8 | 12.90% | 4% | 4% | 4% | 4% | 3% | 8% | 8% | 7% | 5% |
| Coffee | 4.7 | 0.60% | 2% | 2% | 2% | 1% | 2% | 6% | 5% | 3% | 2% |
| Apple | 16.1 | 3.00% | 0% | 0% | 1% | 0% | 0% | 1% | 1% | 1% | 1% |
| Oranges | 17.4 | 3.20% | 1% | 1% | 3% | 0% | 1% | 2% | 2% | 1% | 1% |
| Potato | 69.1 | 13.00% | 3% | 3% | 4% | 3% | 2% | 3% | 3% | 4% | 3% |
| Bread | 39.3 | 7.30% | 3% | 3% | 3% | 4% | 3% | 3% | 3% | 3% | 3% |
| Olive oil | 5.3 | 1.00% | 1% | 1% | 1% | 0% | 1% | 1% | 1% | 1% | 1% |
| Sunflower oil | 5.3 | 1.00% | 4% | 4% | 4% | 4% | 3% | 4% | 4% | 4% | 4% |
| Sugar from beet | 29.8 | 5.50% | 3% | 3% | 4% | 3% | 4% | 2% | 3% | 3% | 3% |
| Milk | 80.1 | 14.80% | 9% | 9% | 8% | 9% | 8% | 8% | 8% | 8% | 8% |
| Cheese | 15 | 2.80% | 15% | 15% | 16% | 15% | 13% | 12% | 13% | 14% | 14% |
| Butter | 3.6 | 0.70% | 7% | 7% | 6% | 8% | 6% | 5% | 5% | 7% | 7% |
| Meat - beef | 13.7 | 2.50% | 18% | 18% | 17% | 18% | 22% | 15% | 16% | 16% | 17% |
| Meat - pork | 41 | 7.60% | 20% | 20% | 16% | 21% | 21% | 19% | 17% | 18% | 19% |
| Meat - poultry | 22 | 4.20% | 8% | 8% | 7% | 8% | 9% | 8% | 8% | 8% | 8% |
| Pre-prepared meal | 2.9 | 0.50% | 1% | 1% | 1% | 1% | 1% | 1% | 1% | 1% | 1% |
| | | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 100% |

[9] **(based on Stranddorf et al. 2005)**

Conversely, the identification of the most relevant impact categories contributing to the overall impact of the entire BoP nutrition presents huge discrepancies when applying different weighting sets (Table 22). This is due to the intrinsic differences among the weighting sets, which build on different perspectives and attribute different weights to the impact categories.

*Table 22 Relevance of the different impact category. Results of the application of several ILCD-compliant weighting sets in combination with normalization references EU27 (Castellani et al., 2016). A color scale is applied to the results in each column (i.e. impact category), from green (lowest contribution), to red (highest contribution).*

| **CONTRIBUTION ANALYSIS** | **1-1-1** | **Castellani et al. (2016) WFsA** | **Castellani et al. (2016) WFsB** | **EDIP2003 (based on Stranddorf et al. 2005)** | **Tuomisto et al., (2012)** | **Bjørn & Hauschild (2015)** | **Bjørn & Hauschild (2015) - global** | **Ponsioen & Goedkoop (2015)** | **Huppes et al. (2012)** |
|---|---|---|---|---|---|---|---|---|---|
| Climate change | 1.9% | 2.0% | 1.5% | 2% | 3.2% | 21.4% | 26.0% | 20.4% | 8.7% |
| Ozone depletion | 0.0% | 0.0% | 0.0% | 2% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% |
| Human toxicity, cancer effects | 6.2% | 6.6% | 5.0% | 7% | n.a. | n.a. | n.a. | 2.2% | 8.1% |
| Human toxicity, non-cancer effects | 44.6 % | 42.5% | 32.3% | 53% | n.a. | n.a. | n.a. | 47.7% | 36.3% |
| Particulate matter/Respiratory inorganics | 2.6% | 3.0% | 2.3% | n.a. | n.a. | n.a. | n.a. | 5.1% | 3.4% |
| Ionizing radiation, human health | 0.5% | 0.5% | 0.4% | n.a. | n.a. | n.a. | n.a. | 0.0% | 0.7% |
| Photochemical ozone formation, human health | 1.0% | 1.2% | 0.9% | 1% | 0.0% | 16.2% | 26.3% | 0.0% | 1.1% |
| Acidification | 7.3% | 8.1% | 6.1% | 9% | 10.1% | 4.7% | 4.2% | 0.3% | 6.1% |
| Eutrophication terrestrial | 8.3% | 8.9% | 6.7% | 10% | 37.9% | 3.1% | 1.8% | 0.0% | 3.9% |
| Eutrophication freshwater | 3.4% | 3.2% | 2.5% | 3% | 3.7% | 13.4% | 4.3% | 0.0% | 1.6% |
| Eutrophication marine | 8.2% | 8.7% | 6.6% | 11% | 37.4% | 5.5% | 4.4% | 0.0% | 3.8% |
| Land use | 2.3% | 2.2% | 1.9% | n.a. | 2.3% | 25.7% | 18.8% | 10.6% | 4.6% |
| Ecotoxicity freshwater | 5.8% | 5.5% | 4.6% | 0% | n.a. | 6.0% | 0.4% | 0.0% | 12.7% |
| Resource depletion water | 6.2% | 5.9% | 28.4% | n.a. | 5.3% | 3.9% | 13.7% | 4.9% | 6.3% |
| Resource depletion, mineral, fossils and renewables | 1.8% | 1.7% | 0.9% | 0% | n.a. | n.a. | n.a. | 8.6% | 2.5% |
| **TOTAL** | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 100% | 100% |

The discrepancies in the results suggest that it is important to acknowledge that different perspectives lead to different results. Moreover, it is relatively straightforward to identify the product groups impacting the most, as from Table 21, when keeping constant normalization references although varying weighing sets.

The authors also checked the most sensitive element between normalization and weighting: their findings indicate normalization as the most sensitive choice if compared

to the selection of a specific weighting set among those considered in their work. However, this result may change in case weighting methods building on different approaches are added to the ones included in their work. Nevertheless, the ranking of products is not significantly affected by changing normalization reference (e.g. from ILCD EU27 to ILCD Global – Laurent et al. 2013). This might be due to few impact categories driving the results regardless of the normalization and weighting methods selected, or to the fact that all impact categories are strongly correlated.

### 5.4.1 Conclusions and recommendations

Weighting sets are often dependent on different value choices to assign different weights to all impact categories. Therefore, different weighting sets lead to significant differences of the final conclusions. It is recommended that the decision about inclusion of which weighting set to use has been made and documented in the first scope definition and shall not be changed later during the study.

# 6 Estimation of uncertainty

Evaluation of the uncertainty of an LCA study is a step which involves both the LCI and LCIA phase, therefore it is strictly linked to the content of the previous chapters. Uncertainty in LCA can be defined in different ways (Ciroth et al., 2016). For examples it can be meant as describing the variability of the LCA data to determine the significance of the indicator results, being "the estimated amount or percentage by which an observed or calculated value may differ from the true value" (Skone, 2000). ISO 14044 defines uncertainty analysis as systematic procedure to quantify the uncertainty introduced in the results of a life cycle inventory analysis due to the cumulative effects of model imprecision, input uncertainty and data variability.

The estimation of uncertainty serves three main purposes:

- it supports a better understanding of the results obtained
- it supports the iterative improvement of an LCA study
- it also helps the target audience to assess the robustness and applicability of the study results

The PEF Guide identifies two key sources of uncertainty in PEF studies:

1. Stochastic uncertainties for the inventory data
   they refer to statistical descriptions of variance around a mean/average. For normally distributed data, this variance is typically described in terms of an average and standard deviation.
2. Choice-related uncertainties
   They arise from methodological choices including modelling principles, system boundaries, allocation choices, choice of impact assessment methods and other assumptions related to time, technology, geographical aspects, etc. These are not amenable to statistical description, but rather can only be characterized via scenario model assessments.

In this report we refer to uncertainty analysis to describe the range of likely outcomes based on a set of inputs and it differs from a sensitivity analysis which is linked to how sensitive the results are to a change in an input parameter (see previous chapters).

It has to be kept in mind that all steps involved in a quantitative process are affected by uncertainties. Therefore, it is important that the uncertainty is assessed:

- At inventory level (e.g. standard deviation associated to a data point). An example on how to assess the uncertainty at inventory level was recently provided by Ciroth et al. (2016)
- At characterization level (e.g. linked to characterization factors)
- At normalisation level (e.g. linked to normalisation factors)
- At weighting level (e.g. linked to weighting factors)

### 6.1.1 Conclusions and recommendations

It is recommended that the overall uncertainty of an LCA study is estimated and reported in the final report. It is also recommended to include an evaluation to which extent the uncertainties identified affect the final conclusions.

# 7 Meta-analysis

The widespread development of LCA in the last decades as a decision supporting tool leads to a huge amount of LCA studies in literature concerning all kind of goods and services ("products"). When trying to make sense of their results that sometimes are very different, or come up with a conclusion, techniques to combine or re-analyze them are needed such as meta-analysis.

Zumsteg et al., (2012) described the key factors for conducting and reporting systematic reviews, including meta-analyses[10], in LCA. Those key factors are simplified in a check list: 1. Review title, keywords, and abstract; 2. Rationale for the review; 3. Review question and objectives; 4. Description of review protocol; 5. Findings and features of the individual studies in the review; 6. Assessment of bias; 7. Synthesis methods (qualitative and quantitative); 8. Limitations of the review; and 9. Summary of findings and conclusions.

This chapter is not intended to give guidelines on how to perform a systematic review or a meta-analysis but to highlight the critical points that the practitioner should take into account when collecting and using the LCA data. Recalling the check list previously described, it is in the findings and features of the individual studies where one of the biggest shortcomings when performing a meta-analysis concerning LCA studies is found. Assembling data from the individual studies is complex due to:

- the existing "flexible" guidance in conducting LCA that leads to different assumptions and methodological choices made in the various LCA modelling exercises. The ISO 14040 and 14044 standards provide the framework for LCA. This framework, however, leaves the individual practitioner with a range of choices, which can affect the legitimacy of the results of an LCA study.
- the lack of homogeneity reporting LCA results.

As found in De Matos et al., (2015) while doing a meta-analysis of different bioeconomy value chains, the most influencing differences among studies mainly relate to:

- The definition of the system boundaries and the stages included in the study (e.g. even if the same general system boundaries are considered - e.g. cradle to gate - some studies may or may not include intermediate transport, construction and decommissioning of buildings, etc.).
- The impact assessment methods used, as different methods may consider, for example, different substances for a given impact category, and different characterization factors for the same substance.
- The definition of the functional unit (e.g. as the input, the output product, the agricultural land unit, etc.) (Cherubini and Stromman, 2011).
- The consideration of direct and indirect land use change (dLUC and iLUC, respectively) (Cherubini and Stromman, 2011).
- The technology considered in the process and its maturity level.
- The approach used to model the multifunctional system. For instance, if substitution is used, the reference system selected may have a significant influence on the final LCA results. On the other hand, if allocation is used, the selection of the allocation criteria and the relative

[10] The definition of Meta-analysis given by Zumsteg et al., (2012) is "A melding of data from multiple studies, usually involving additional mathematical analysis, with the goal of utilizing this synergy of information and data size to answer questions that cannot be answered by existing individual studies or to improve the certainty or impact of know findings by increasing the sample size. Meta-analyses are often performed as part of a systematic review"

contribution of each co-product may considerably influence the results of the assessment.

These points are in line with those reported in other studies in literature. Wolf et al., (2015) conducted a meta-analysis of LCAs for wood energy services (heat, power and combined heat and power (CHP)) and chose a set of "decisive parameters" to be analyzed arguing that those factors promote inconsistent findings along LCA studies. Apart from the previously mentioned, the authors include the following factors (some of them specific of the good or service under study):

- Data sources such as empirical data, literature, well-known databases, unspecified databases, simulators or expert interviews are some of them.
- Transportation distances and types
- Feedstock properties (in this case wood)
- Combustion capacity and efficiency and co-combustion rates
- Energy service provided (power, heat, and CHP)

Schreiber et al., (2012) performed a meta-analysis on LCA studies of electricity generation with carbon capture and storage (CCS) comprising the three main CCS technologies (post-combustion, oxyfuel and pre-combustion). In line with previous studies, they highlighted the different choices in the LCA methodology and parameters (specific of the good or service under study) that heavily impact the overall results. Here we report those which differ from the already mentioned:

- Data quality and availability
- Time horizon as most studies considered present and future power plants as far as 2050.
- Spatial representation – some elements in the CCS process chain are highly site-specific such as storage sites.
- Life cycle inventory (LCI) – which inputs and outputs are considered.
- Operational valuation and weighting methods
- Power plant efficiency and energy penalty of the capture process
- Carbon dioxide capture efficiency and purity
- Fuel origin and composition

Moreover, when having a look at the limitations of the review and meta-analysis some studies just focus on GHG emissions. Wolf et al., (2015) set the CML midpoint impact category Global Warming as the basis for the comparison of results because it is represented in 100% of the studies. They did not consider other less-frequent impact categories, but yet important, such as acidification, eutrophication, and particulate matter, owing to the lack of results published in the studies. On the other hand, Schreiber et al., (2012) not only focused on GHG but they included other impacts such as acidification potential, eutrophication potential, photochemical ozone creation potential and cumulative energy demand.

As a conclusion, increased harmonization and transparency in the calculation and reporting of LCA data and results is needed (Cristobal et al., 2015).

### 7.1.1 Conclusions and recommendations

The number of LCA studies available is increasing and making sense of all the results that can be obtained from them is a difficult task. As a conclusion increased harmonization and transparency in the calculation of LCA data and results is needed. It is recommended that reporting of LCA data is done in a more consistent and harmonized

manner, to allow users of LCA study a better understanding of the results and conclusions drawn.

# 8 Conclusion

One of the key aims of LCA is to provide the decision makers with comprehensive and understandable information: this task is achieved by a proper interpretation of the results of an LCA study.

This report builds on the requirements provided in relevant international standards (ISO 14044) and guides (ILCD Handbook (EC-JRC, 2011a), PEF Guide (EC, 2013b) and it proposes a practical approach on how to perform a robust interpretation of an LCA study. It provides a ready-to-use approach for the identification of significant issues, and it includes examples of completeness, sensitivity and consistency checks based on real-case studies, which are used to draw the attention of the reader on how to evaluate the results of an LCA study.

The scheme proposed in this report should not be intended as a comprehensive list of items to check, instead it should be seen as a normal workflow that should be carried out by an LCA practitioner in order to derive robust conclusions out of the LCA study. More accurate and case-specific checks should be done depending on the intended goal and scope of the study.

The report is structured in order to allow a practitioner focus on the two quantitative phases of LCA: the life cycle inventory (LCI) and the life cycle impact assessment (LCIA). Regarding the LCI, the report goes through the analysis of the data sources, by giving examples on how to interpret the system boundaries and the foreground system); it provides an overview on data quality and it comprises further completeness and consistency checks (e.g. cut-off, long term emissions, anomaly checks).

When analysing the LCIA the report provides insights on sensitivity analysis regarding characterization, normalization and weighting. When dealing with characterization, the attention of the reader is drawn also on mapping between the inventory and the characterization models and a discussion on the evaluation of uncharacterized elementary flows is also included.

Finally, an overarching topic, such as meta-analysis was deemed useful to be included as this is often used when there is the need of assessing the state of the art of the LCA studies on a specific product/ supply chain. Key messages stemming from this report are as follows:

- Results reveal that the relative importance of one substance compared to another changes with the characterization model. This is because the models differ in the substances that they include and in the CFs of each substance. It is recommended to run a sensitivity check with different characterization models to evaluate the robustness of the conclusions drawn. It is also recommended to evaluate the theory underlying the characterization models chosen to evaluate a certain impact category, as this helps understanding the results obtained.
- Results of the LCIA phase can be highly affected by the characterization model chosen. Furthermore, the evaluation of uncharacterized elementary flows is of primary importance to assess the completeness of the LCIA coverage of the identified LCI flows. Indeed, a high number of elementary flows inventoried might be non-characterized by the chosen models, even if their relevance in the assessed impact categories could be of primary importance. Therefore, it is highly recommended that a quantification of the uncharacterized elementary flows is reported before any interpretation or conclusion is performed on the superiority of one product versus another.
- Different normalisation sets may produce different results in terms of identification of most relevant impact categories, life cycle stages and processes. Different reference systems refer to different situation which can all be valid,

depending e.g. on the geographical scope of the study. It is recommended to evaluate different normalisation tests, which fit the study at hand, to support the conclusions drawn with the baseline scenario or to provide additional insights.

- Weighting sets are often dependent on different value choices to assign different weights to all impact categories. Therefore, different weighting sets lead to significant differences of the final conclusions. It is recommended that the decision about inclusion of which weighting set to use has been made and documented in the first scope definition and shall not be changed later during the study.
- It is recommended that the overall uncertainty of an LCA study is estimated and reported in the final report. It is also recommended to include an evaluation to which extent the uncertainties identified affect the final conclusions.
- The number of LCA studies available is increasing and making sense of all the results that can be obtained from them is a difficult task. As a conclusion increased harmonization and transparency in the calculation of LCA data and results is needed. It is recommended that reporting of LCA data is done in a more consistent and harmonized manner, to allow users of LCA study a better understanding of the results and conclusions drawn.

Further aspects, such as the appropriateness of the functional unit, the intended goal, the system boundaries and value choices should normally be part of an interpretation, in order to identify recommendations and limitations of the study and draw the appropriate conclusions. They are not discussed further in this report, but it is recommended that at least, each decision based on a personal or a stakeholder's value shall always be documented and explained in the conclusions of the LCA study. This is consistent with one of the key aims of LCA, which is to provide the decision makers with comprehensive and understandable information.

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## List of abbreviations and definitions

| | |
|---|---|
| AOX | adsorbable organic halogens |
| BOD | biochemical oxygen demand |
| BTEX | benzene, toluene, xylene isomers |
| CFs | characterisation factors |
| COD | chemical oxygen demand |
| DQR | data quality requirements |
| EcF | ecological footprint |
| EF | environmental footprint |
| ElFl | elementary flow |
| HMs | heavy metals |
| IC | impact category |
| ILCD | International Reference Life Cycle Data System |
| LCA | life cycle assessment |
| LCI | life cycle inventory |
| LCIA | life cycle impact assessment |
| OEFSR | organisation environmental footprint sector rules |
| PAH | polycyclic aromatic hydrocarbons |
| PEF | product environmental footprint |
| PEFCR | product environmental footprint sector rules |
| PPP | plant protection products |
| OEF | organisation environmental footprint |
| TAB | technical advisory board |
| TS | technical secretariat |
| UNEP/SETAC LCI | Unep Setac Life cycle Initiative |
| VOC | volatile organic compounds |
| WEEE | waste electrical and electronic equipment |

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