Chapter 9
The Link Between Life Cycle Inventory Analysis and Life Cycle Impact Assessment

Jutta Hildenbrand and Rickard Arvidsson

Abstract In this chapter, the link between life cycle inventory analysis (LCI) and life cycle impact assessment (LCIA) is discussed. For the feasibility of conducting a life cycle assessment (LCA) and for making its results more robust, it is necessary that data collected in the LCI stage are suitable for the LCIA methods, and in particular for comparative studies, it is relevant to provide matching levels of detail for all compared options. Four illustrative examples are provided: (i) the differences in receiving compartment resolution for toxic emissions, (ii) differences in stressor resolution for particulate matter formation, (iii) lacking characterization factors for metal use, and (iv) lacking characterization factors for sum parameters and not fully specified emissions (such as BOD, TOC and "alkanes, unspecified"). Two important lessons to consider for maintaining a strong link between LCI and LCIA are highlighted based on these examples. First, it is suggested that it is important to have the same resolution between LCI data and LCIA methods. Scenario analysis, where different resolutions are assumed and tested, can be a strategy in cases where differences in resolutions are unavoidable. Second, ways to handle the absence of characterization factors are discussed, including the development of additional characterization factors that match the available LCI data and derivation of characterization factors from process information.

Keywords Life cycle assessment (LCA) · Life cycle impact assessment (LCA) · Life cycle inventory analysis (LCI) · Metal scarcity · Particulate matter · Toxicity · Sum parameters · Unspecified information

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1 Introduction

A life cycle inventory analysis is often described as the most time-consuming part of a life cycle assessment (LCA) study, mainly because it is linked to collecting and validating data. An underlying assumption is then that ready-made life cycle impact assessment (LCIA) methods can be used with the collected information and that the inventory data is fit for the chosen assessment method (Guinée 2015). This means that it is vital to have guidance on what data to search for and to be able to use the data in the subsequent LCIA. One vital part of this is what data could later be purposefully used in the selected LCIA methods. If the LCIA methods selected do not include impacts from a certain stressor – or if no LCIA method at all includes an impact that is potentially linked to an emission – there is no point in searching for inventory data on emissions of that stressor from an LCA perspective. However, it could be argued that data for emissions are published in environmental reports due to a potential environmental impact, even if the cause-effect chain is not completely clear, therefore, when emission or resource demand information is available it should be considered on principle. Conversely, if there is no inventory data for a certain type of emissions – such as acidifying emissions – there is no point in including LCIA methods for impacts caused by that type of emissions. There is thus a clear link between LCI and LCIA, which follows naturally from the equation used to calculate inventory data into impact scores (*IS*) (Hauschild and Huijbregts 2015):

$$IS_j = \sum_i \sum_k \sum_l Q_{i,k,l} CF_{j,i,k,l} \tag{9.1}$$

In Eq. 9.1, CF stands for characterization factor, Q for the quantity of emission or resource use, i is a certain contributor to the impact category j, k is the location of the emission or resource use, and l is the environmental compartment to which the emission occurs or from which the resource is extracted. If no CF for the contributor i exist, there is little point in gathering data on Q for that contributor. And conversely, if there exist no data on Q for a contributor, and if it is not possible to derive data based on estimates, such as from emission factors for an individual process, the existence of a CF for the contributor is of little help. Consequently, LCI and LCIA should preferably be linked to one another, and fit together like two pieces of a puzzle (Fig. 9.1).

Such mismatches between LCI and LCIA can be of vital importance in comparative LCAs. Assume that two products A and B are being compared. For A, available data on resource use and emissions is possible to match with CFs from

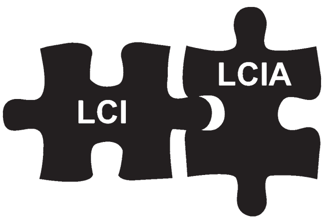

Fig. 9.1 Ideal illustration of how life cycle inventory analysis (LCI) and life cycle impact assessment (LCIA) should preferably fit together

contemporary LCIA methods. But for B, CFs might be missing for several emissions, or the resolution of the LCI data might not match that of the CFs. For example, the LCI data for B might contain many "toxic emissions to water," but as will be shown in Sect. 2, contemporary toxicity impact assessment methods need a specification regarding which type of water (fresh- or seawater). If those emissions related to B are then excluded and A receives a higher impact, it will remain unclear if B truly has lower impacts than A, or if the result is an artifact stemming from mismatches between LCI data and LCIA methods. As a short summary, for comparative studies, it is highly relevant to provide symmetry of LCI data regarding resolution and completeness.

This chapter is thus about the link between LCI and LCIA. This link is something that should be reflected upon in the goal and scope of the study, as well as iteratively throughout the whole LCA study and in particular while performing sensitivity analysis. Four illustrative examples of potential mismatches between LCI and LCIA are shown – in other words, examples when LCI and LCIA do not necessarily fit as nicely together as the illustration in Fig. 9.1 suggests. The purpose of these examples is to highlight pitfalls that can arise in an LCA study. The chapter concludes with some recommendations that will hopefully contribute to the reader being able to avoid such pitfalls in the future.

2 Receiving Compartment Resolution: The Example of Toxic Emissions

The emission of toxicants to the environment is an important problem – indeed, many historical risks that spurred regulatory responses were related to chemicals such as polychlorinated biphenyls and tributyltin (Harremoës et al. 2001). Other current chemicals risk causing severe impacts to human health and the environment include, for example, cadmium (Järup and Åkesson 2009) and endocrine disruptors (Bergman et al. 2013). Impacts from such emissions can range from local damage to ecosystem collapse and are clearly of relevance to include in LCA.

The current consensus model for assessing toxicity impacts in LCA is called USEtox, which has been developed as part of the Life Cycle Initiative of the United Nations Environment Programme and the Society for Environmental Toxicology and Chemistry (Hauschild et al. 2008; Rosenbaum et al. 2008). The currently most recent version of USEtox – version 2.01 – can be found on its webpage usetox.org, along with information and documentation about the underlying model. Still in use is also the USES-LCA 2.0 model (Huijbregts et al. 2000), which is the base for toxicity assessment used in the most recent version of the LCIA package ReCiPe from 2016 (Huijbregts et al. 2016; Huijbregts et al. 2017) and also included in the baseline set proposed by CML for midpoint indicators (Guinée et al. 2002). These two methods differ in a number of regards. For example, the measurement units for the toxicity impacts in USEtox are increased morbidity cases per kg substance for human toxicity and PAF m^3 day per kg substance for ecotoxicity, where PAF stands for potentially affected fraction of species. For USES-LCA, the measurement unit for both human toxicity and ecotoxicity is 1,4-DCB equivalents, where DCB stands for the reference substance dichlorobenzene. The USES-LCA model includes three different ecotoxicity impact categories (freshwater and marine water ecotoxicity potential as well as terrestrial ecotoxicity potential). The authors of USEtox chose to include only freshwater ecotoxicity as an impact category due to lack of experimental data for terrestrial and marine ecosystems (Rosenbaum et al. 2008).

Another notable difference is that they consider emissions to different environmental compartments. USEtox considers emissions to the eight compartments indoor air, industrial indoor air, urban air, continental rural air, continental freshwater, continental seawater, continental natural soil, and continental agricultural soil. USES-LCA considers emissions to the five compartments air, freshwater, seawater, agricultural soil, and industrial soil. By "consider," we here mean that CFs are provided for emissions to these compartments. See Fig. 9.2 for an illustration of the coverage of the two models. USEtox consequently has a higher resolution regarding the air compartment, whereas the two models have the same resolution for the water compartment. Regarding the soil compartment, they both include agricultural soil, but USEtox includes natural soil and USES-LCA includes industrial soil.

The question for LCA practitioners is whether the compartment resolutions of these two models map with available inventory data? Some data – in particular old data or data found in non-LCA databases and sources, including environmental and sustainability reports based on mandatory data required by supervisory

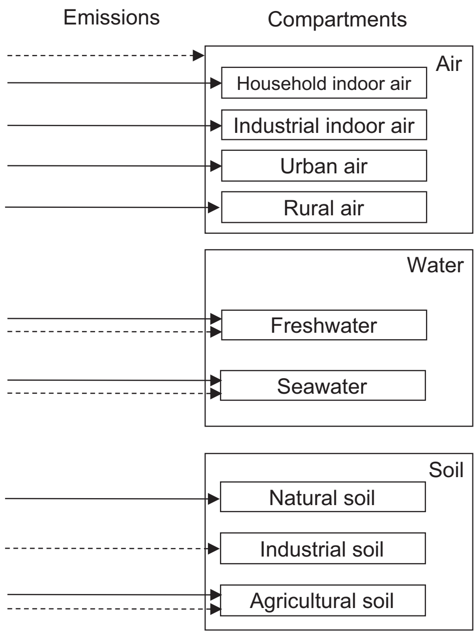

Fig. 9.2 Illustration of the coverage of the USEtox model (continuous lines) and the USES-LCA model (dashed lines) regarding emissions to different compartments

authorities – might not have specified the receiving compartment at all but only note that an emission occurs. In other cases, there might be a receiving compartment reported (or possible to deduce), but the resolution of the compartment is limited. For example, emissions from road transport can be modeled and calculated based on fuel use and emission factors, for example, in models based on the Handbook Emission Factors for Road Transport (HBEFA), developed by INFRAS (Switzerland), which are currently used in Germany, Austria, Switzerland, Norway, and Sweden and supported by the Joint Research Centre (JRC) of the European Union. Air is the natural receiving compartment for such combustion-related emissions, but as noted above, the USEtox model also requires information about whether the air is urban, rural, household indoor, or industrial indoor. Clearly, the two first are the only ones of relevance for traffic-related emissions, but the HBEFA does not specify whether emissions occur in urban or rural areas. The choice of specific receiving compartment and corresponding characterization factor might have an influence on the results, for example, by a factor of 47 as shown in Table 9.1. For an LCA study using USEtox, such unspecified emission data should therefore preferably be assigned more specific receiving compartments based on other information sources.

Also, in other cases, the practitioner may need to make a qualified assumption about the receiving compartment given some knowledge about the emission, for example, assuming that volatile substances are emitted into air. Data inventoried in the often-used LCA database Ecoinvent database (2013) are clearer regarding receiving compartment, but generally only report emissions to three aggregated compartments: air, soil, and water. Whether that air is urban, rural, or even indoor is unknown, so is whether the water is fresh- or seawater or soil is industrial or agricultural. The higher resolution of the USEtox and USES-LCA models is then of little help to the (many) users of the ecoinvent database. In general, newer LCIA methods that were developed after data were published in databases could not be considered when the data were first collected, and it is not always possible to provide more detailed information with data updates.

It is perhaps easy to think that the higher resolution a model has, the better it is. However, as discussed in Sect. 1, without LCI data to fit the LCIA model, the assessment will become hindered. Providing a higher resolution in the LCIA model can thus paradoxically result in problems for the conducting of the LCA and perhaps even result in less comprehensive LCA results. For example, if emissions to soil are available in the LCI data and a low-resolution LCIA method provides CFs for

Table 9.1 Midpoint-level human toxicity characterization factors with two significant numbers from the USEtox 2.01 model for urban and rural air, respectively, using the example of tetrachloroethylene (CAS RN: 127–18-4)

Emission compartment	Characterization factor, cancer + non-cancer [10^{-7} CTUh/kg]
Urban air	0.18
Rural air	8.5

(unspecified) soil only, there is a possibility to conduct the assessment using Eq. 9.1. However, if the LCIA method provides CFs for more specified soil compartments, such as natural or agricultural soil, then the LCI data and the LCIA model do not fit together. As mentioned above, it may in some cases be possible to make a qualified assumption about a likely compartment based on available information. In other cases, one possible way to handle such a situation of resolution mismatch could be through different what-if scenarios (Höjer et al. 2008). For example, one can assume in one scenario that all soil emissions are to natural soil, and that all soil emissions are to agricultural soil in another scenario. However, if there is a large mismatch between the LCI data and LCIA models, that would result in quite many scenarios, making the interpretation step more challenging.

3 Stressor Resolution: The Example of Particulate Matter Formation

Particulate matter (PM) is an air pollutant harmful to human health, which has been reported to cause roughly three million deaths annually (WHO 2016). The damaging PM fractions are particles with diameters below 10 μm, called PM10, since they can penetrate deep into the lungs. Furthermore, smaller particles than that, with diameters below 2.5 μm, called PM2.5 are even more damaging than larger-sized PM. In LCIA, particulate matter has sometimes been included as part of the human health impact category, as in the EDIP2003 (Hauschild and Potting 2003) and USES-LCA (Huijbregts et al. 2000) methods. More recently, it has become more common to view it as a separate impact category. The question then becomes: Which particles to consider? In particular, which particle size fraction should be considered? Less than 2.5 μm, less than 10 μm, or something else?

In the ReCiPe 2008 impact assessment method, PM10 is considered (Goedkoop et al. 2013), but in the newer version of ReCiPe from 2016, PM2.5 is considered instead (Huijbregts et al. 2016). Both the IMPACT 2002+ (Humbert et al. 2012) and IMPACT World+ (Bulle et al. 2019) impact assessment methods use PM2.5. The recent recommendation from the UNEP/SETAC life cycle initiative is also an impact assessment method based on PM2.5 as input (Frischknecht and Jolliet 2016). There thus seems to be a tendency toward preferring PM2.5 over PM10.

The question is how this focus on PM2.5 in contemporary LCIA methods match available inventory data? The ecoinvent database reports particle emission data in three size ranges: (i) <2.5 μm, thus corresponding to PM2.5, (ii) >2.5 μm and < 10 μm, thus corresponding to PM10 minus PM2.5, and (iii) >10 μm, which corresponds to particle sizes not considered harmful to human health. The inventory data corresponding to the lowest of these ranges thus match the CFs reflecting PM2.5 well. The sum of the first two match CFs reflecting PM10. The second and third ranges by themselves have no CFs available that match them.

There thus seems to be a good potential for a match between LCI data and LCIA methods for PM formation, but practitioners should be aware that particle emission data can be reported for different sizes. A careful matching of particle emission data and CFs is therefore important. In cases where they do not match, there might be ways around that. For example, it is stated in the user manual for the IMPACT 2002+ method that inventory data is often available for PM10 rather than PM2.5 (Humbert et al. 2012). To account for this, they recommend to use the relationship that the PM2.5 content of PM10 in air is approximately 0.6. The PM10-based inventory data can then be multiplied by 0.6 as a correction factor to reflect PM2.5 emissions instead. Similarly, they write that the PM2.5 share of the total particular matter (PM_{tot}) is approximately 0.33, which can be used to correct inventory data reporting the total particulate matter. These correction factors can be used in cases when the LCIA method considers PM2.5 but the inventory data is reported as PM10 or PM_{tot}.

4 Missing Characterization Factors: The Example of Metal Use

Metals are important raw materials to many life cycles. For many products, the inventory consists of a considerably long list of metal input flows. For some products, the use of metals is the perhaps highest concern of the inventory. One example of this is electric vehicles, where scarcity of metals required for the batteries, such as lithium, has been reported to be a major concern for their future use (Kushnir and Sandén 2012). Another example is the use of tellurium, gallium, ruthenium, and silver that may limit the development of solar cells (Tao et al. 2011). Yet another example is liquid-crystal displays, where the use of indium is making the screens more expensive, which has spurred the development of alternative transparent and conductive materials produced from less scarce materials (Arvidsson et al. 2016). Although not all products have metal use as the most pressing issue, most products would probably have some sort of metal input in a comprehensive inventory.

Unfortunately, there is no consensus on how to assess metal use in LCA. Klinglmair et al. (2014) conducted a review of existing LCIA models for assessing abiotic resource depletion and found that there was a lack of consensus on underlying principles, but also a lack of consensus regarding which metals should have the highest abiotic resource depletion potential. For example, for one method (the CML 2002 method), aluminum was considered two orders of magnitude less impacting than iron. For another method (the EPS 2000 method), aluminum was considered two orders of magnitude more impacting than iron. Such inconsistencies point toward a potential for further development of methods for assessing metal use. Such work will require the specification of what is actually meant by metal scarcity – for instance, in terms of timeframe (Drielsma et al. 2015).

Another observation by Klinglmair et al. (2014) was that some LCIA models included many different metals, whereas others considered only a few (Table 9.2).

Table 9.2 List of life cycle impact assessment (LCIA) models for assessing metal use

Metal	Exergy	CML 2002	EI 99	EDIP 97	EPS 2000	IMPACT 2002+	ReCiPe
Aluminum	x	x	x	x	x	x	x
Antimony		x		x	x		
Arsenic		x			x		
Barium		x			x		
Beryllium		x		x	x		
Bismuth		x			x		
Cadmium		x		x	x		
Chromium	x	x	x	x	x	x	x
Cobalt		x		x	x		x
Copper	x	x	x	x	x	x	x
Gallium		x			x		
Germanium		x			x		
Gold		x		x	x		x
Indium		x			x		
Iron	x	x	x	x	x	x	x
Potassium		x			x		
Lead	x	x	x	x	x	x	x
Lithium		x			x		
Magnesium		x					
Manganese	x	x	x	x	x	x	x
Mercury		x	x	x	x	x	
Molybdenum	x	x	x	x	x	x	x
Nickel	x	x	x	x	x	x	x
Niobium		x			x		
Palladium	x	x		x	x		x
Platinum	x	x		x	x		x
Rhenium	x	x			x		
Selenium		x		x	x		
Silicon		x					
Silver	x	x		x	x		x
Sodium		x					
Strontium		x		x	x		
Tantalum		x		x	x		
Tellurium		x		x	x		
Thallium		x		x	x		
Tin	x	x	x	x	x	x	x
Titanium		x		x	x		
Tungsten		x	x	x	x		
Vanadium		x		x	x		
Yttrium		x		x	x		
Zinc	x	x	x	x	x	x	x
Zirconium		x		x	x		
Total	**14**	**42**	**12**	**29**	**39**	**11**	**15**

Which metals they include are marked by "x." Obtained from Klinglmair et al. (2014)

Take, for example, the six scarce metals mentioned above: lithium, tellurium, gallium, ruthenium, silver, and indium. Lithium, gallium, and indium are only included in two models (CML 2002 and EPS 2000). Tellurium is included in three models (CML 2002, EDIP 97 and EPS 2000). Silver is included in five models (Exergy, CML 2002, EDIP 97, EPS 2000, and ReCiPe). Ruthenium seems not to be included in any of the reviewed models. The CML 2002 and EPS 2000 methods are the two most inclusive methods.

This lack of coverage by some abiotic resource depletion LCIA models constitutes a challenge for linking LCI to LCIA. Assume that an assessor has an inventory list with high input amounts of a seldom-included metal. She then faces the obvious choice between (i) using one of the few LCIA models that do include the metal or (ii) leave it out from the LCIA step, only report it in terms of LCI results and maybe discuss its resource impacts qualitatively. The first alternative is problematic, since the LCIA models for abiotic resource depletion are based on different principles and it is not certain that the few models that include e.g., indium are based on the principles most suitable for the study as a whole. The second alternative is also problematic, since resource impacts from one of the main inputs then remain unassessed quantitatively. There could, however, be a third alternative in some cases. Some of the LCIA models for resource depletion provide equations that can be used by the assessor to calculate additional CFs. For example, the CML 2002 method applies the Eq. 9.2 (Guinée et al. 2002):

$$ADP_i = \frac{DR_i / R_i^2}{DR_{ref} / R_{ref}^2} \tag{9.2}$$

where DR is the extraction rate (kg/year), R is the reserve of the resource (kg), i is the resource assessed, and "ref" stands for a reference material, which is antimony in the CML 2002 model. Based on this equation, the assessor can, often without too much trouble, calculate CFs for any metal she may want to assess – for instance, ruthenium. She may even alter the equation in order to adapt to other overarching assessment principles. For example, Drielsma et al. (2015) suggested that the crustal content of a metal resource may be more relevant for long-term decisions than are reserves. The provision of such equations enables the assessor to improve the coverage of the LCIA models to match that of the LCI data.

There is a fourth alternative that may be applicable in some cases, which is again to use what-if scenarios. If there is a certain input of a scarce metal for which the CF is not known and cannot be calculated based on the LCIA model chosen, a worst-case scenario may be employed in order to investigate whether that metal input could constitute a notable share of the resource impact. In the worst-case scenario, it can be assumed that the metal's abiotic resource depletion potential is equal to the highest known for any metal in that LCIA model. If, then, the input metal becomes dominant, it is a sign that its resource impacts should be further investigated. This alternative is mainly relevant when the metal with unknown depletion potential constitutes a minor part of the total metal input. If it is the dominating input on an LCI basis, multiplication with the highest known CF will only emphasize this dominance.

5 Missing Characterization Factors: The Example of Sum Parameters

Data that have to be reported to supervisory authorities are sometimes also included in the environmental and sustainability reporting of companies, in particular over series of several years that show development over time. These data are based on sum parameters that are suitable for routine measuring such as adsorbable organic halides (AOX), chemical oxygen demand (COD), biochemical oxygen demand (BOD), total suspended solids (TSS), and total organic carbon (TOC) for waste water. For emissions to air, reported data include volatile organic compounds (VOC), among others. Sum parameters are often also used as a basis in environmental permits and can be used to evaluate the technical efficiency of treatment and remediation techniques. However, sum parameters are rarely processed in LCIA methods, even though they clearly are proxies for environmental impacts. One of the reasons is that substances with different CFs are included in a sum parameter, and without further information it is not possible to identify a specific stressor. An overview of the few instances of including sum parameters in ready-made LCIA methods is shown in Table 9.3.

COD is established in several (older) methods, often with comparably low CF values. More recent methods that include CFs for COD include Ecological Scarcity, EPS, and the North American method TRACI, for which CFs are lower than for any other flow that contributes to the respective categories.

Ecological Scarcity considers regulatory emission limits and goals for Switzerland as a basis to calculate Eco-Factors and is, therefore, able to accommodate even sum parameters for which no single stressor is identified (Frischknecht and Büsser Knöpfel 2014). The categories “water pollutants” and “non-radioactive waste” included in the Ecological Scarcity method do not refer to any specific impact, but that is not required for regulatory limits to be established. This allows for flexibility in the Ecological Scarcity method regarding the inclusion of sum parameters. No other method has been identified to account for AOX and TOC.

While sum parameters are not often considered in LCIA methods, they certainly indicate that emissions occur. The level of detail in LCIA methods requires, however, more specific information that can be related to a substance or compound. Where this information is available, considering sum parameters additionally can lead to the overestimation of impacts. However, information based on single species can also be incomplete. For example, Köhler (2006) noted that the toxicity of specified individual organic contaminants in waterborne organic emissions could not

Table 9.3 Examples of sum parameters in selected LCIA methods

Sum parameter	Method	Impact category
COD (water)	Ecoindicator 95 v2.1	Eutrophication
COD (water)	CML 2002 baseline	Eutrophication
COD (water)	IMPACT 2002+	Aquatic eutrophication
COD (water)	Ecological Scarcity 2013	Eutrophication
COD (water)	TRACI v2.1	Eutrophication
COD (water)	EPS 2015	Fish and meat production capacity, Species extinction
BOD5 (water)	TRACI v2.1	Eutrophication
BOD5 (water)	EPS 2015	Fish and meat production capacity, Species extinction
AOX (water)	Ecological Scarcity 2013	Water pollutants
VOC	ILCD 2011 Midpoint v1.10	Photochemical ozone formation
VOC	IMPACT 2002+	Respiratory organics
TOC (water/groundwater long-term)	Ecological Scarcity	Non-radioactive waste to deposit

explain the ecotoxicological impacts of the emission as observed based on toxicity tests. The effluent potentially contained additional organic substances that were not listed individually but contributed to the TOC. She therefore instead attempted to develop CFs for the whole TOC parameter by considering the general fate and effects of TOC in wastewater treatment, resulting in broad ranges due to the lack of detailed data on TOC mixtures. Despite the broad CFs ranges, applying them in a scenario fashion is an attractive alternative to merely omitting the impacts of reported TOC emissions completely. Similar attempts could be made for other sum parameters and/or other industry branches than wastewater treatment.

Creating CFs specific for sum parameters is thus one strategy to create a match between LCI data and LCIA methods. Another strategy is to disaggregate sum parameters so that they fit existing CFs for more specific stressors. This requires knowledge regarding emission sources and processes. Where sum parameters are the only information available for process emissions, a first approach could then be to identify sector-specific emissions based on which substances are used in the process or, where this is not available, based on literature sources. The sum parameter can then be disaggregated into sub-components or, if the composition is not well known, different scenarios reflecting different compositions can be tested. For example, where emissions are listed as “(mineral) oils, unspecified, to river,” this will not be considered in an LCIA method, despite the fact that mineral oils contain toxic organic stressors such as polycyclic aromatic hydrocarbons (Almeda et al. 2013) and inorganic stressors including a large variety of heavy metals (Fedorov et al. 2007). Further specification of the composition of oils is recommended to make sure that emissions from sources are evaluated and considered in the assessment. Information for that is, however, unfortunately not widely published, and triangulation based on several data sources needs to be carried out. The approach helps to overcome data gaps, but is still based on estimates and assumptions.

6 Discussion and Outlook

Based on the four examples above, two important lessons for linking LCI and LCIA can be identified. The first is about the resolution in the LCI data and the LCIA model. As shown for the cases of receiving compartments for toxic emissions and particulate matter fractions, a resolution mismatch can create problems for conducting an LCA study. In some cases, the practitioner may be able to make qualified assumptions to increase the resolution of the LCI data to match the LCIA model. This could, for example, be done for PM by correcting for the content of PM2.5. If not possible, different scenarios, where, for example, it is assumed that all toxic emissions to water occur to freshwater, might provide hypothetical cornerstone results. Given LCI data with low resolution, the practitioner might need to revisit the selection of LCIA methods and instead chose one with similarly low resolution. And the other way around – if the LCI data has a higher resolution than the LCIA method, an LCIA method with higher resolution might instead be selected.

We also recommend that developers of LCIA models provide recommendation on how to handle situations with more or less aggregated LCI data, e.g., if emission data for to water is available but the model requires emission data for freshwater specifically. A general note that can be made is that since LCI data acquisition is often expensive and time-consuming, it is recommendable to develop LCIA models with available LCI data in mind.

The second lesson is the importance of available CFs, as shown for the cases of metal use and sum parameters. A problematic situation with lacking CFs for important emissions and recourses is difficult to resolve in any satisfactory way. As an advanced option, a practitioner can follow Köhler (2006) and develop new CFs that match the available LCI data. As a less advanced option, scenarios might again provide some guidance. Assigning worst-case CFs to metals lacking specific CFs might tell whether the metal has any potential to constitute a hotspot in the assessment.

The examples in this chapter clearly show that problems can arise when there is a mismatch between LCI analysis and LCIA. To facilitate the application of LCA, such mismatches should therefore preferably be avoided. The examples provided here, and the two main lessons about the importance of similar resolution and availability of CFs can hopefully help reducing such mismatches in the future.

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