Chapter 2
Introduction to Life Cycle Assessment

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This chapter[1] gives an overview of the mainstream method of life cycle assessment (LCA) on the basis of the generally accepted principles as laid down in International Organization for Standardization (ISO) series of Standards on LCA. The first part is devoted to the key questions addressed by LCA and sketches the historical development towards that method. The second part provides an overview of the LCA method itself, while the third part discusses some examples of LCA applications. Finally, the fourth part discusses some of the future challenges to LCA including life cycle sustainability assessment (LCSA) and streamlined LCA techniques.

2.1 LCA: What Does It Address and How Has It Developed?

Today's society is highly consumption based. Through advertisements and marketing campaigns, we are stimulated on a daily basis to consume products, services and, if possible, more and more. Products and services are the key selling items of our economic system. At the same time we are facing huge sustainability challenges with respect to, e.g. climate change, land use change, water shortages, toxic pollution and resource scarcity. Products and services are key concepts in addressing these sustainability challenges. Refrain from any consumption is not an option, so we have to strive for a more sustainable production and consumption pattern. Environmental

[1] This chapter is partly based on Heijungs, R. & Guinée, J.B. (2012). An Overview of the Life Cycle Assessment Method—Past, Present, and Future. In: Curran, M.A. (Ed.), Life Cycle Assessment Handbook: A Guide for Environmentally Sustainable Products, pp. 15–42. Beverly: Scrivener Publishing.

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policy today focuses at the transition to such sustainable production and consumption patterns. This is taking place in various ways and at various levels.

For this purpose we often will have to compare the sustainability of products against each other. We then first need a few definitions: what is sustainability, what are comparable products and what exactly are products?

- Sustainability has at least three dimensions: the economic, the environmental and the social dimensions; LCA focuses on the environmental dimensions only.
- Products fulfil functions or services. We buy a car, motor or bike to travel from A to B; we eat and drink to satisfy our nutritional needs; etc. We generally have different alternative products to fulfil these functions.
- We are not so much looking into products as such but rather at a system of economic or industrial processes needed for the functioning of that product. Here the term "product system" (or even better "function system") enters the arena. A product system refers to the entire life cycle of a product, from extraction of natural resources to final waste management of the disposed product, from "cradle to grave". We don't just look at driving a car, but we include the total system of industrial/economic processes needed for constructing all components of the car, the maintenance of the car, the use of the car including the total life cycle of the fuels needed to drive the car, the roads needed, etc. until the final disposal of the car at the end of its life which may include recycling.

Knowledge of the environmental impacts of such product systems is indispensable if we are aiming for improving the environmental performance of these systems. We preferably need numbers for all relevant environmental impacts of product systems, from the cradle to the grave (whole life cycle), in order to find best solutions for their improvement without shifting impacts to other fields or to other phases of the life cycle (trade-offs). One of the assessment methods widely used for this is environmental life cycle assessment, abbreviated LCA.

Do we really need a potentially complex method as LCA? Isn't driving an electric vehicle simply better than driving a gasoline-based vehicle? If it was as simple as that, we wouldn't need LCA but unfortunately the reality is much more complex if we take a systems approach mapping the whole life cycle and all potentially relevant environmental impacts. Electric vehicles need a lot of batteries to store electricity, and these batteries need a lot of scarce resources that may potentially leak to the environment as hazardous substances, but depending on the source of electricity, they may perform better in the use phase compared to gasoline-based cars. Only by analysing the full life cycle of these two functionally equivalent "car driving systems", we can determine which one is environmentally better performing. But even then, it's often not possible to simply conclude that one option is better than the other as it may still depend on consumer's behaviour or on the source of electricity (e.g. renewables or fossil fuels) as in the case of the electric car (Hawkins et al. 2012). The same ratio can be applied to comparing plastic and carton disposable mugs with porcelain mugs for drinking coffee, disposable diapers versus cotton diapers, biofuel- versus fossil fuel-based electricity systems, etc.

2.1.1 LCA in a Nutshell

LCA offers a method for quantitatively compiling and evaluating the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle (ISO 1996), and its results may be used to support decision-making in this area. LCA refers to a method, but it also refers to the result of this method. In this chapter, we will mainly focus on the method that is used to obtain an "LCA result".

LCA has made a long way, and it is still changing. But since a decade or so, there is a broadly accepted set of principles that can claim to be the present-day LCA framework based on a series of standards and technical reports issued by the ISO, the 14040 series (ISO 1996, 1998, 2000, 2000a, b, 2002, 2003, 2006, 2012, 2012a, b). This series consists of the documents listed in Table 2.1.

The standards are organized into the different phases of an LCA study. These are:

- Goal and scope definition
- Inventory analysis
- Life cycle impact assessment
- Life cycle interpretation

The relations between these phases have been illustrated in a figure, and this figure has become a sort of logo of LCA (Fig. 2.1).

Table 2.1 ISO documents on LCA

Number	Type	Title	Year
14040	International Standard	Principles and framework	1996, 2006
14041	International Standard	Goal and scope definition and inventory analysis	1998[a]
14042	International Standard	Life cycle impact assessment	2000a[a]
14043	International Standard	Life cycle interpretations	2000b[a]
14044	International Standard	Requirements and guidelines	2006[b]
14047	Technical report	Examples of application of ISO 14042	2003
14047	Technical report	Examples of ISO 14044 impact assessment application	2012a
14048	Technical specification	Data documentation format	2002
14049	Technical report	Examples of application of ISO 14041	2000
14049	Technical report	Examples of ISO 14044 goal and scope definition and inventory analysis application	2012b
14067	Technical report	Carbon footprint of products; requirements and guidelines	2012

[a]Updated in 2006 and merged into 14044

[b]Replaces 14041, 14042 and 14043

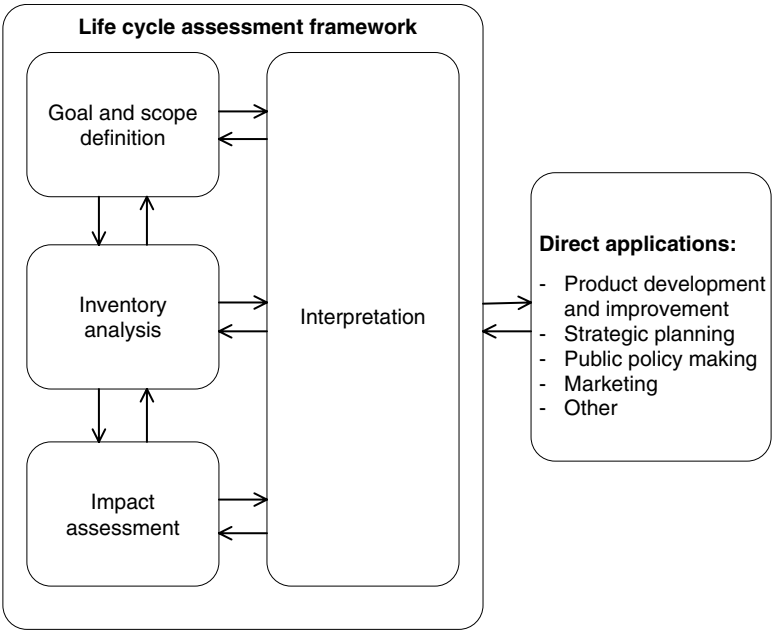

Fig. 2.1 The general methodological framework for LCA (ISO 1996)

2.1.2 History of LCA

As mentioned, LCA has made a long way. This section discusses the historical development of LCA in terms of its past and present. The text is largely based on Guinée et al. (2011).

2.1.2.1 The Past of LCA (1970–2000)

Firstly, we will briefly discuss and evaluate LCA as developed and applied in the past while distinguishing two periods: (1) 1970–1990 and (2) 1990–2000.

1970–1990: Decades of Conception

The first studies that are now recognized as (partial) LCAs date from the late 1960s and early 1970s, a period in which environmental issues like resource and energy efficiency, pollution control and solid waste became issues of broad public concern (Assies 1992). One of the first (unfortunately unpublished) studies

quantifying the resource requirements, emission loadings and waste flows of different beverage containers was conducted by Midwest Research Institute (MRI) for the Coca Cola Company in 1969. Together with several follow-ups, this marked the beginning of the development of LCA as we know it today. After a period of diminishing public interest in LCA and a number of unpublished studies, there has been rapidly growing interest in the subject from the early 1980s on. In 1984 the Swiss published a report (BUS 1984) that presented a comprehensive list of the data needed for LCA studies, thus catalysing a broader application of LCA.

The period 1970–1990 comprised the decades of conception of LCA with widely diverging approaches, terminologies and results. There was a clear lack of international scientific discussion and exchange platforms for LCA. During the 1970s and the 1980s, LCAs were performed using different methods and without a common theoretical framework. LCA was repeatedly applied by firms to substantiate market claims. The obtained results differed greatly, even when the objects of the study were the same, which prevented LCA from becoming a more generally accepted and applied analytical tool (Guinée et al. 1993).

1990–2000: Decade of Standardization

The 1990s saw a remarkable growth of scientific and coordination activities worldwide. Also the first scientific journal papers started to appear in the *Journal of Cleaner Production*, in *Resources, Conservation and Recycling*, in the *International Journal of LCA*, in *Environmental Science & Technology*, in the *Journal of Industrial Ecology* and in other journals.

The Society of Environmental Toxicology and Chemistry (SETAC) started playing a leading and coordinating role in bringing LCA practitioners, users and scientists together to collaborate on the continuous improvement and harmonization of LCA framework, terminology and methodology. Next to SETAC, the ISO has been involved in LCA since 1994. Whereas SETAC working groups focused at development and harmonization of methods, ISO adopted the formal task of standardization of methods and procedures.

The period of 1990–2000 can therefore be characterized as a period of *convergence* through SETAC's coordination and ISO's standardization activities, providing a standardized framework and terminology, and platform for debate and harmonization of LCA methods. In other words, the 1990s was a decade of standardization. During this period, LCA also became part of policy documents and legislation. The main focus was on packaging legislation.

Although this decade is mainly one of convergence, it is also the stage of scientific scrutiny, research into the foundations of LCA, and exploring the connections with existing disciplines. For instance, we observe sprouting ideas on consequential LCA and related allocation methods (Weidema 2000, Ekvall 2000) marking the transition to the present decade of LCA.

2.1.2.2 2000–Present: The Era of Elaboration

The first decade of the twenty-first century has shown an ever increasing attention to LCA. In 2002, the United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC) launched an International Life Cycle Partnership, known as the Life Cycle Initiative (http://www.lifecycleinitiative.org/). In 2005 the European Platform on Life Cycle Assessment (http://eplca.jrc.ec.europa.eu/) was established to promote the availability, exchange and use of quality-assured life cycle data, methods and studies for reliable decision support in (EU) public policy and in business. In the USA, the US Environmental Protection Agency started promoting the use of LCA (http://www.epa.gov/nrmrl/std/lca/lca.html), and also US environmental policy got increasingly life cycle based all over the world (e.g. http://www.gpo.gov/fdsys/pkg/BILLS-110hr6enr/pdf/BILLS-110hr6enr.pdf).

In this same period, several life cycle-based carbon footprint standards have been, or are being, established (http://www.lcacenter.org/LCA9/special/Carbon-footprint.html). Although the footprint concept is broader than LCA (it, for instance, includes organization footprints and national footprints as well), a rich literature has spawned around the product environmental footprint (PEF; http://ec.europa.eu/environment/eussd/smgp/product_footprint.htm), which basically is an LCA, and in addition carbon footprints (CF), which are LCAs focusing on greenhouse gases only. CF is now often associated with consumer products, and in an increasing number of cases, we find the CF score on the packaging or the label. The introduction of carbon footprint standards for supporting policies and (bioenergy) performance-based regulation raised some severe problems, which have often not yet been solved adequately (Matthews et al. 2008). Life cycle-based carbon footprint results need to be robust and "lawsuit-proof". This implies that the freedom of methodological choices for the handling of, e.g., biogenic carbon balances and allocation should be reduced to an absolute minimum and that uncertainties are dealt with appropriately, which is unfortunately not yet common practice. Another issue is the limited scope of carbon footprints towards climate change impacts neglecting the possible trade-off to other impact. Finally, we reiterate that the term footprint not necessarily refers to a functional unit-based LCA; see also Hoekstra's (2017), Chap. 4 in this volume.

The period 2000–present can be characterized as the decade of elaboration. While the demand on LCA increases, the period is characterized by a *divergence* in methods again. As ISO never aimed to standardize LCA methods in detail and as there is no common agreement on how to interpret some of the ISO requirements, diverging approaches have been developed with respect to system boundaries and allocation methods, dynamic LCA, spatially differentiated LCA, environmental input–output-based LCA (EIO-LCA) and hybrid LCA, which combines the strengths of classical LCA and environmentally extended input–output analysis (EE-IOA). On top of this, various life cycle costing (LCC; cf. Hunkeler et al. 2008) and social life cycle assessment (S-LCA; cf. Benoît and Mazijn 2009) approaches have been proposed and/or developed.

2.2 Overview of LCA

Typically, LCA starts by defining goal and scope, then proceeds to the inventory analysis, then optionally continues to impact assessment and ends with the interpretation. However, as indicated in Fig. 2.1, an LCA study is a highly iterative process, so that the LCA practitioner may need to go back to goal and scope after the preliminary inventory work, to move back from impact assessment to inventory analysis, to have a look at the interpretation in an early stage, etc.

Below, we will discuss the main idea and content of the four phases distinguished in Fig. 2.1 in separate subsections. All quotations are taken from the ISO documents, unless otherwise indicated.

2.2.1 Goal and Scope Definition

There is no explicit ISO definition of the first phase of LCA. However, it obviously centres around formulating the question and stating the context of answering this question. In the goal and scope definition, no data is collected and no results are calculated. Rather, it is a place where the plan of the LCA study is defined as clearly and unambiguously as possible. Likewise, in an LCA report, it should help the reader to quickly find out the precise question addressed and main principles chosen.

The goal of the LCA should deal with the following topics:

- The intended application
- The reasons for carrying out the study
- The intended audience
- Whether the results are to be used in comparative assertions disclosed to the public

The choices made here have an influence on the rest of the LCA procedure. For instance, depending on the intended audience, a critical review may be needed, and it may be important that an external expert takes this task.

In the scope definition, a number of major choices are made. First of all, the product system or systems to be studied and the function the system delivers (or in case of a comparative LCA, the functions the systems deliver). For instance, one might be interested in the product systems' incandescent light bulb versus the LED bulb, with the function of lighting a room.

An important aspect of the scope definition is the functional unit. It is obviously pointless to compare an incandescent bulb with a LED bulb: the life spans and performances differ considerably, and the function is not having a light bulb but having light of a certain quality. The functional unit expresses the function of the products and thereby offers a way to equalize differences in performance. A functional unit for analysing lighting systems could thus better be phrased in terms of the function,

for instance "lighting a standard room of 15 m^2 with 1000 lm for 1 h". As LCA employs mathematically a linear calculation rule, the results will scale by choosing a numerically different functional unit (say, "lighting a standard room of 20 m^2 with 800 lm for 3 h"), but the alternatives considered will scale up or down consistently, so this will not affect the conclusions. A consequence is, however, that LCA cannot tell if a product is "sustainable" or "environmentally friendly"; LCA can only indicate if product X is "more sustainable" or "more environmentally friendly" than product Y or that the use phase is the "least sustainable" or "least environmentally friendly" part of the life cycle for product Z.

The scope definition further sets the main outline on a number of subjects that are discussed and further refined in more detail in the later phases. These include, amongst others:

- System boundaries
- Impact categories
- Treatment of uncertainty

The ISO standard and some other texts at places suggest that these topics are implemented in detail in the scope definition. This is wrong: the goal and scope definition is not concerned with collecting data or calculating results, so no concrete details on such topics can be specified at this phase.

2.2.2 Inventory Analysis

ISO defines inventory analysis (LCI) as the "phase of life cycle assessment involving the compilation and quantification of inputs and outputs for a product throughout its life cycle". It will be clear that quantification is an important aspect here, and numbers, in terms of data and calculations, are of central concern in the inventory analysis.

The LCI is built up on the basis of the unit process. A unit process is the "smallest element considered in the life cycle inventory analysis for which input and output data are quantified". Examples of unit process are coal mining, steel production, refining of oil, production of furniture, use of a TV, recycling of wastepaper and transport by lorry. Each of these processes is described in quantitative terms as having inputs and outputs. As a matter of fact, a unit process is in LCA considered as a black box that converts a bundle of inputs into a bundle of outputs. Inputs come in several types: products (including components, materials and services), waste for treatment and natural resources (including fossils, ores, biotic resources and land). Outputs come in several types as well: again products (including components, materials and services), waste for treatment and residuals to the environment (including pollutants to air, water and soil, waste heat and noise); see Fig. 2.2.

Unit processes form the building blocks of an LCA. This is because products are not harmful for the environment as such, except for the processes involved in

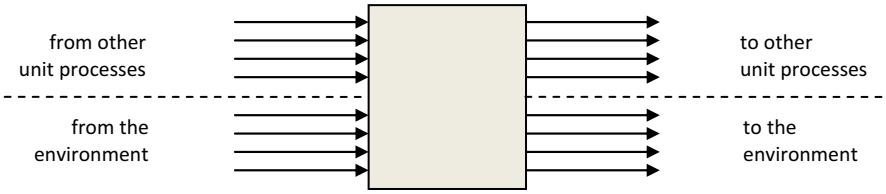

Fig. 2.2 General template of a unit process. The process (*grey rectangle*) is considered as a *black box*, having inputs (*left-hand side*) and outputs (*right-hand side*) from and to other unit process (*top lines*) and from and to the environment (*bottom lines*)

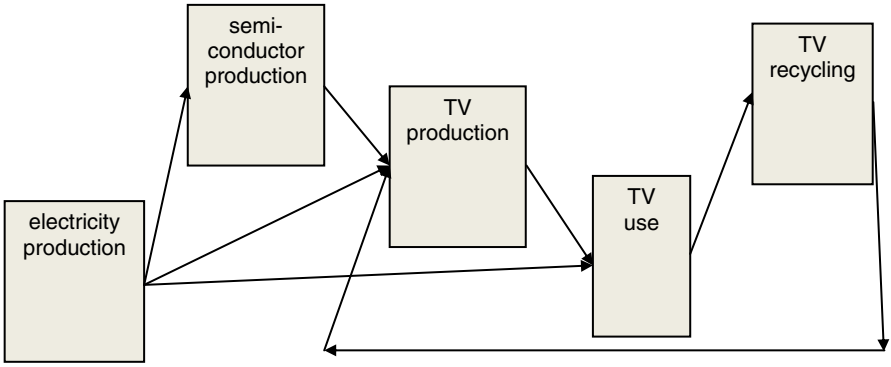

Fig. 2.3 Fragment of a simplified flow diagram for an LCA on television sets. Because the purpose is to show how unit process are connected, only the flows from and to other unit processes are displayed, and flows from and to the environment are hidden. All transport, packaging, etc. have been left out as well

products. Producing, using and disposing products create the burden to the environment. Therefore, these processes assume a central position in LCA.

The essential feature of LCA in which it distinguishes itself from the analysis of an industrial or agricultural process is that it connects different unit process into a system. A flow diagram is a graphical representation of the system of connected unit processes. Figure 2.3 shows a fragment of such a flow diagram.

As we can see, some unit processes are connected with one another in simple upstream–downstream connections, e.g. TV production is upstream connected to semiconductor production. But there are also more complicated connections, e.g. electricity linking to different parts of the system and recycling feeding back to production. Flow diagrams are in fact huge webs of interconnected unit processes. In the present era of digital databases, LCA studies can easily comprise several thousands of unit processes.

LCA is primarily a quantitative model. In the LCI, all unit processes included have to be quantified. This means that we have to specify the sizes of the inflows and outflows, per unit process. As an example, let us take the unit process of aluminium production.

Table 2.2 Example of a unit process specification

Type of flow	Name	Amount	Unit
Inputs from other unit processes	Aluminium oxide	2	kg
Inputs from other unit processes	Electricity	20	kWh
Outputs to other unit processes	Aluminium	1	kg
Outputs to the environment	CO_2	0.2	kg

An aluminium plant may specify their technology in terms of inputs and outputs by stating input requirements (e.g. 2 kg aluminium oxide and 20 kWh electricity per kg produced aluminium) and emissions (e.g. 200 g CO_2 per kg produced aluminium). We must translate this into our template for unit processes; see Table 2.2.

For each of the unit processes included, quantitative data should be collected. Moreover, in order to be able to process the data and perform the calculations automatically, a clear and unambiguous representation is needed. This implies, amongst others, harmonization of nomenclature (e.g. not using "carbon dioxide" for one unit process and "CO_2" for another one) and harmonization of units (e.g. not mixing up kg and lbs).

In Table 2.2, the unit process data is given per unit of output, here per kg of aluminium. In an LCA, we must next find out how much we need. For instance, the product may need 3 kg of aluminium, not 1 kg. The basic assumption of the LCA model is that technologies are linear. This means that we can scale the data of a unit process by a simple multiplication. In the example, 3 kg of aluminium would require 6 kg of aluminium oxide and 60 kWh of electricity, while it would release 600 g of CO_2. The assumption of linear technology is an important restriction of LCA; yet it is an important step in making the calculation and data collection feasible.

In scaling the unit processes, the web-like nature of the system quickly creates complications, as everything depends upon everything. The calculation of the scaling factors, and with that of the emissions to and extraction from the environment, is greatly simplified by considering the problem as a system of linear equations: one unknown (the scaling factor) for every unit process and one equation (a balance) for every flow. Thus, solutions may be obtained by matrix algebra. The details of this are not discussed here; see (Heijungs and Suh 2002) for a detailed exposition.

The approach mentioned above may fail in a number of cases. We mention two complications:

- For some products, upstream production processes or downstream disposal process may be difficult to quantify.
- For some unit processes, the balance equations become impossible due to the fact that these processes produce not just one product but several products.

The first issue can be solved by a procedure known as cut-off, the second one by allocation.

Cut-off is a solution to the problem that the system is theoretically infinitely large. To produce a TV, we need machines, and these machines are produced by machines, and these machines in turn need machines, etc. But of course we have an

intuitive idea that some very distant upstream processes will be quite unimportant. This means that we will cut off certain inputs: although we know that something is needed, and we sometimes even know how much is needed, we do not go into the trouble of specifying how these inputs are produced. It turns out to be difficult to specify reliable criteria when cut-off is allowed or to estimate how large the error is when a cut-off is made. Criteria on the basis of negligible contribution to mass or cost (e.g. smaller than 1 %) often work pretty well but occasionally have been shown to yield large errors. Alternatively, estimates of missing parts by means of similar processes (e.g. estimating production of a freezer by production of a fridge) or by economic input–output tables may be helpful. Another approach is to do a difference analysis: in comparing a CRT and an LCD TV, we may leave out the broadcasting processes.

The second problem has given rise to one of the biggest controversies in LCA theory. The problem can be stated simply: if a transportation process needs gasoline, the upstream unit process is a refinery that produced not only gasoline but also diesel, kerosene, heavy oils and some more. The direct impacts (from pollutants like CO_2), but also the flows to and from other processes that may lead to impacts (e.g. from oil drilling), may be argued not to be attributed to gasoline only but in need to be distributed over gasoline, diesel and all other coproducts. This is hardly contested, but the debate focuses on how to do this. To make it more concrete: how much of the CO_2 from a refinery is allocated to the gasoline? Different schools have provided different arguments, and none of these have been completely compelling so far. Some solutions lead to strange results, other solutions may be very difficult to carry out (e.g. for lack of data or appropriate software), and still others are rejected by a large number experts. To complicate the issue, the problem does not only occur in unit processes that produce several coproducts but also in unit processes that treat more than one type of waste or that recycle a waste into a good. It is even not agreed if the multi-output case, the multi-input case and the recycling case must be treated in the same way or not.

Within ISO, a preference order for solving the multifunctionality problem has been designed. It distinguishes several solutions (dividing the unit process into two or more subprocesses, expanding the system to include the additional functions, partitioning on the basis of a physical parameter, partitioning on the basis of an economic parameter) separated by clauses like "wherever possible" and "where … cannot be established". This stepwise procedure is a clear comprise, and in practice it leaves so much freedom that LCA studies that are according to the ISO standard can give conflicting results. One peculiarity deserves to be mentioned: besides the ISO-based "expanding the system to include the additional functions", we often see a method that is best described as "subtracting the avoided impacts from additional functions" but that is more commonly known as the substitution method or the avoided burdens method. For instance, when a waste treatment activity co-produces electricity, the emissions from the regular way of producing the same amount of electricity are subtracted. This method has similarities with that of system expansion, but of course it is not identical. Many LCA studies employing the substitution method claim to be ISO compliant, even though strictly speaking

ISO 14044 does not mention this method, let alone recommend it. That does not necessarily mean that these studies are incorrect, of course. Compliance with ISO is not a sufficient quality guarantee but also not a necessary one.

After appropriate cut-off and allocation steps, the final inventory results can be calculated. Typically, this is a table with the quantified inputs from and outputs to the environment, for each of the alternative systems considered, expressed in relation to the functional unit. With the present-day software and databases, this inventory table may be 1000 lines long or more. It contains not only the familiar pollutants and resources, such as CO_2, NO_x and crude oil, but also more exotic items, such as 1-pentanol, cyprodinil and dolomite. Typically, these so-called elementary flows are aggregated over the entire system, so that the CO_2 number is the life cycle emission of CO_2.

2.2.3 Impact Assessment

Life cycle impact assessment (LCIA), or impact assessment in short, is the "phase of life cycle assessment aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts for a product system throughout the life cycle of the product". Its motivation comes from two observations:

- The final result of the inventory analysis, the inventory table, is too long (e.g. 1000 different items) to handle.
- The inventory table contains many items that require expert knowledge (such as 2-methyl-2-butene) to understand in terms of importance.

Impact assessment, and in particular the characterization step, solves both issues: it "involves the conversion of LCI results to common units and the aggregation of the converted results within the same impact category".

While the unit process is the central element of the inventory analysis, the central element in impact assessment is the impact category. ISO defines it as a "class representing environmental issues of concern to which life cycle inventory analysis results may be assigned". Perhaps more helpful are some examples: climate change, toxicity and depletion of fossil energy carriers.

As climate change (often used interchangeable with global warming) is a well-known issue, we will illustrate the main ideas of impact assessment with this case. The inventory table contains a number of greenhouse gases: CO_2, CH_4, N_2O, etc. These are known to contribute all to the phenomenon of climate change. Climate change involves long sequence of causal mechanisms: emissions of greenhouse gases lead to changes in the composition of the atmosphere, which lead to a change in the radiation balance, which in turn leads to a change in the temperature distribution, which leads to changes in climate, which leads to changes in ecosystems and human activities, etc. The further we proceed in this causal chain, the more uncertain and speculative our knowledge becomes. While quite some scientific evidence is available with respect to the composition of the atmosphere, the impacts on

biodiversity are debated. Many of these later impacts are even conditional on our future activities, including future emission scenarios and mitigating actions. To be able to quantitatively model the emissions of different greenhouse gases into an impact indicator for climate change, we must do several things.

First, we must choose a certain point in the causal mechanism. This can be at the front end (change in radiation balance), at the back end (change of biodiversity) or somewhere in between (change in temperature). In LCA, two main schools have emerged:

- Those that focus on the front end, the so-called midpoint approach.
- Those that focus on the back end, the so-called end-point approach.

The midpoint approach has the advantage that it includes fewer debatable assumptions and less-established facts; the end-point approach has the advantage that it provides more intuitive metrics (like loss of life years instead of kg CO_2 equivalent). Regardless of the choice between midpoint and end point, the indicator chosen is referred to as the impact category indicator or category indicator in short.

Second, a way must be found to convert the emission data into the chosen impact indicator. Scientists in chemistry, meteorology, ecology, etc. have developed model fragments to estimate the atmospheric lifetimes of greenhouse gases, their effect on the radiation balance and the formation of clouds, the effects of temperature on the distribution of species, etc. These fragments have been combined by workgroups from the UN-based Intergovernmental Panel on Climate Change (IPCC) into quantitative models of the impacts of greenhouse gas emissions. Part of this are the global warming potentials (GWPs), which are quantitative measures of the strength of different greenhouse gases. Many midpoint LCIA methods apply GWPs for climate change. We will illustrate their usage below. For now, it suffices to mention that GWPs provide one example of a set of characterization factors and that the IPCC model from which they are derived is an example of a characterization model. Note, by the way, that IPCC has not developed this model as a characterization model for LCIA but that the LCA community has adopted this model as such and its derived GWPs as characterization factors. Also note that the characterization model itself is not used by LCA practitioners; only the characterization factors that have been derived from it as a one-time exercise are used. Characterization factors are often tabulated in LCA guidebooks and are implemented in many LCA software packages, while the characterization models often require supercomputers and expert knowledge.

In fact, there is one element before one can select a category indicator and a characterization model with associated characterization factors. It is the selection of impact categories to be addressed. Some LCA studies concentrate on just one impact category. For instance, the carbon footprint (of a product, not of a company or country) is a form of LCA that addresses just climate change at the midpoint level through GWPs. (Some chapters in this book go into more depth on a single impact category: see Chap. 3 by Boukherroub et al. (2017) for more on carbon footprinting, Chap. 4 by Hoekstra (2017) for more on water footprinting and Chap. 5 by Blass et al. (2017) for more on depletion of material resources.) At the other extreme, some LCA studies incorporate 15 or more impact categories. For consistency

Table 2.3 Overview of widely used impact categories with examples of category indicators at midpoint and end-point levels

Impact category	Midpoint category indicator	End-point category indicator
Climate change	Infrared radiative forcing	Loss of life years, fraction of disappeared species
Ozone layer depletion	Change in tropospheric ozone concentration	Loss of life years
Acidification	H^+ concentration	Fraction of disappeared species
Eutrophication	Biomass potential	Fraction of disappeared species
Human toxicity (sometimes split into carcinogens, noncarcinogens, respiratory effects, etc.)	Time-integrated exposure, corrected for hazard	Loss of life years
Ecotoxicity (sometimes split into aquatic toxicity, terrestrial toxicity, marine toxicity, etc.)	Time-integrated exposure, corrected for hazard	Fraction of disappeared species
Depletion of energy carriers	Primary energy requirement	Decreased availability
Depletion of material resources	Amount of material used, corrected for availability and/or importance	Decreased availability
Land use impacts	Amount of land occupied or transformed	Fraction of disappeared species
Water use impacts	Amount of water used or displaced	Decreased availability

reasons, the choice of impact categories is often made on the basis of a recommended impact assessment guidebook or its implementation in software. Thus, in practice one often sees LCA studies reporting the use of "IMPACT2002+", "TRACI", "CML-IA", "ReCiPe", "ILCD", etc. All these methods comprise a recommended set of impact categories with a category indicator and set of characterization factors. ISO does not specify any choice in these matters. Table 2.3 gives an overview of some often-used impact categories and category indicators. We see that the column with end-point indicators contains many times the same term (e.g. "loss of life years"). This suggests that impact categories can be aggregated into fewer end-point indicators than midpoint indicators.

As a concrete example of how characterization works, let us study a fragment of a hypothetical inventory table, containing the following information: emission of CO_2 100 kg, emission of CH_4 1 kg and emission of SO_2 1 kg. Characterizing greenhouse gases with GWPs requires a table with GWPs. In such a table, one can find that the GWP of CO_2 is 1 (by definition) and that the GWP of CH_4 is 25 (kg CO_2 equivalent/kg CH_4). SO_2 has no GWP; it is assumed not to contribute to climate change. Characterization now proceeds in the case of climate change by calculating

$$1 \times 100 + 25 \times 10 = 350\,\text{kg}\,CO_2 \text{ equivalent}$$

For the more general case, this can be written as

$$GW = \sum_s GWP_s \times m_s$$

where GW is the global warming score, s the substance (the different greenhouse gases), GWP_s the GWP of substance s and m_s the emitted amount of substance s in kg. This may be further generalized as

$$I_c = \sum_s CF_{c,s} \times m_s$$

where c codes for the impact category, I_c represents the indicator result for category c and $CF_{c,s}$ is the characterization that links substance s to impact category c. This formula is the operational formula for characterization. With a table of characterization factors specified, it makes clear:

- That LCIA builds on the results of LCI (as is clear from the term m_s)
- That characterization converts the results of LCI into a common metric (as is clear from the multiplication by CF)
- That characterization aggregates the converted LCI results (as is clear from the summation symbol)

The results from characterization are a list of numbers, for instance, a score for climate change, a score for toxicity, etc. ISO refers to such numbers as "category indicator results", but most LCA practitioners prefer names like "score", sometimes expanded with the name of the impact (like in "toxicity score"). The complete list is known by names like "LCIA profile", "characterization table", etc.

An optional next step is normalization referring to calculating "the magnitude of the category indicator results relative to some reference information". It is an optional step for ISO, and indeed, many LCIA studies stop at the characterization. The reference information is in most cases that total impact in a certain region in a certain time period, e.g. in the country of decision in 1 year. Normalization is done "to understand better the relative magnitude for each indicator result". Without normalization, the indicator results are in quite different units, e.g. kg CO_2 equivalent for climate change and MJ primary energy for fossil energy depletion. To put these results in perspective, the normalization expresses them as a share of the total impact size in the region. Arbitrary differences due to a choice of units disappear, and it becomes clear to which impact category a product contributes relatively much. The units of the normalize indicator results are equal; nevertheless such numbers cannot meaningfully be added because the severity of the different impact categories has not yet been accounted for. This can be done in the weighting step; see below. Normalization fulfils several functions: it provides insight into the meaning of the impact indicator results, it helps to check for errors, and it prepares for a possible weighting step.

Weighting is a final step of the impact assessment phase. Weighting, like characterization, converts and aggregates, but while characterization does so for the LCI results, weighting starts with the characterization (or normalization) results. Typically, weighting factors are applied, either to the characterization indicator results or to their normalized version. The weighting factors themselves are supposed to reflect value judgements, such as social and political priorities. Weighting typically produces one final number, by means of

$$W = \sum_c WF_c \times I_c$$

where I_c again symbolizes the impact score (or normalized impact score) for impact category c, WF_c the weighting factor for this impact category and W the weighted result. Well-known examples of such weighted results are the eco-indicator and the ELU (environmental load unit).

2.2.4 Interpretation

ISO defines the interpretation as the "phase of life cycle assessment in which the findings of either the inventory analysis or the impact assessment, or both, are evaluated in relation to the defined goal and scope in order to reach conclusions and recommendations". Several elements are mentioned by ISO:

- Identification of significant issues
- An evaluation that considers completeness, sensitivity and consistency checks
- Conclusions, limitations and recommendations
- Appropriateness of the definitions of the system functions, the functional unit and system boundary
- Limitations identified by the data quality assessment and the sensitivity analysis

The text of ISO on interpretation is very concise, and no details are given on procedures and techniques to be employed. The same applies to most guidebooks on LCA. They mention carrying out an uncertainty analysis but give no clear guidance on how this should be done.

In another context, we have introduced the distinction between procedural and numerical approaches (Heijungs and Kleijn 2001):

- Procedural approaches include all types of analyses that deal with the data and results in relation to other sources of information, like expert judgements, reports on similar products, intuition, reputation of data suppliers and so on.
- Numerical approaches include those approaches that somehow deal with the data that is used during the calculations, without reference to those other sources of information, but as algorithms that use and process the data in different ways, so

as to produce different types of "smart" data reduction that provide an indication of reliability, key issues, discernibility, robustness and so on.

This distinction helps to understand some important roles of interpretation. On the one hand, it is about comparing the data and results with previous findings and to put the results in the context of decision-making and limitations. On the other hand, it is devoted to a systematic analysis with the help of statistical and other decision-analytic techniques. The latter type may be incorporated in software, and indeed, an increasing number of software packages contain options for running Monte Carlo analysis, doing sensitivity analysis, carrying out statistical significance tests, etc. For instance, in the CMLCA software, we have implemented, amongst others:

- Contribution analysis
- Comparative analysis
- Uncertainty analysis
- Perturbation analysis
- Key issue analysis
- Discernibility analysis

The iterative nature of the ISO framework (Fig. 2.1) shows up in this context. Whenever the uncertainties are too high, we may go back to collect better data. Whenever sensitivity analysis shows that some decisions are crucial, we may go back and do a more refined analysis. In this way, the interpretation helps to prepare for a balanced decision but also helps to improve the LCA.

The development of methods in this area is booming (see Henriksson et al. 2013; Henriksson et al. 2015), but current practice is quite meagre, unfortunately. We still see many LCA studies without uncertainty or sensitivity analysis, even though methods and software increasingly facilitate this. There is of course a psychological argument that a contractor pays for finding out something, not for increasing the doubt. And as many LCA practitioners spend several months on collecting data, it is never a nice thing to waste this effort in a last-minute uncertainty analysis. But decision-making obviously means also taking into account the limits of knowledge. Moreover, as discussed before, a proper analysis of uncertainties and sensitivities helps to prioritize the steps earlier on in the framework: collecting data, setting boundaries and making choices.

2.2.5 LCA in Practice

In the text above, the emphasis has been on the generally accepted practice. This is a mix of the ISO standards and a not precisely defined set of guidebooks (e.g. Wenzel et al. 1997; Guinée et al. 2002; Baumann and Tillman 2004; ILCD 2010; Curran 2012; Klöpffer and Grahl 2014). All these texts interpret, add, refine or modify the ISO standards. As has been indicated at a few places, the practice in LCA is sometimes different from what the ISO standards prescribe. There are differences in

terminology (e.g. one seldom sees the term intermediate products), in method (cf. the frequent use of the substitution method), in quality control (judged by frequent absence of uncertainty analyses), etc.

There are also de facto additional standards, dictated by the use of software (for an overview, see http://eplca.jrc.ec.europa.eu/ResourceDirectory/toolList.vm) and databases (for an overview, see http://eplca.jrc.ec.europa.eu/ResourceDirectory/databaseList.vm). Many software packages for LCA have built-in options for impact assessment and uncertainty analysis, but nearly always in a restricted form, allowing some variants and prohibiting other variants. LCI databases are often constructed with predefined allocation methods and cut-off rules, so the user cannot choose otherwise and cannot carry out sensitivity analyses.

2.3 Examples of LCA Applications

LCA has been applied to a wide range of products and services. Until the late 1990s, bibliographies of LCA case studies performed were kept up to date (Grotz and Rubik 1997) on a continuous basis. Using Internet search machines results in a list of hundreds of LCA case studies documented in scientific papers or reports. Even more studies have been made for company-internal purposes, without publication in the scientific literature or on the web. LCA has been applied to simple products as shopping bags and packaging to more complex products such as mobile phones, PCs, cars and buildings. Studies may involve both an environmental comparison between existing products but also the development of new products (ecodesign). LCA has also been applied to services such as LCAs on hazardous waste site cleanup options, on waste management strategies and on different modes of freight transport (road, rail, water). As in the case of product LCAs, it is the function provided which is the core object of these service LCAs, but in this case the function is cleaning up a hazardous waste site, waste management or freight transport.

The results of these case studies were often in line with general expectations, but there were also numerous counter-intuitive results. We randomly provide a selection of examples of the latter below.

Fargione et al. (2008) stirred the biofuel debate by introducing the concept of "biofuel carbon debt". The increasing demand for biofuels was initially increasing the production of biofuels from food crops such as corn, sugarcane, soybeans and palms. As a result, land in undisturbed ecosystems, especially in the Americas and Southeast Asia, was being converted to biofuel production as well as to crop production (indirect land use change) when existing agricultural land was diverted to biofuel production (direct land use change). This land clearing releases huge amounts of CO_2 as a result of burning or microbial decomposition of organic carbon stored in plant biomass and soils over a long time. Fargione et al. (2008) called the amount of CO_2 released during the first 50 years of this process the "carbon debt" of land conversion. Over time, biofuels can afterwards repay this carbon debt if their production and combustion have less net GHG emissions compared to the life

cycle emissions of the fossil fuels they displace. Their conclusion was that "converting rainforests, peatlands, savannas, or grasslands to produce food crop–based biofuels in Brazil, Southeast Asia, and the United States creates a 'biofuel carbon debt' by releasing 17–420 times more CO_2 than the annual greenhouse gas (GHG) reductions that these biofuels would provide by displacing fossil fuels. In contrast, biofuels made from waste biomass or from biomass grown on degraded and abandoned agricultural lands planted with perennials incur little or no carbon debt and can offer immediate and sustained GHG advantages". As policies throughout the world were increasingly promoting biofuels, this publication significantly influenced reconsidering these policies.

Bovea et al. (2010) compared the environmental life cycle performance of different alternatives for the management of municipal solid. They analysed and evaluated 24 different waste management scenarios. They concluded that "fuel consumed during the collection, transport and waste sorting stages makes a contribution to the impact in all the categories that were analyzed, since there is not any avoided environmental burden attributable to those processes". Moreover they conclude that "recycling allows the pollution burden to be avoided for all impact categories, since it avoids the consumption of virgin material according to the substitution rate of 1:1; the contribution made by landfilling depends on whether it is carried out with or without energy recovery […]". This study shows the dependency of results of methodological choices such as the choice of allocation approach, which is particularly a problem in waste management studies (see also Ekvall et al. 2007).

Hawkins et al. (2012) developed a very comprehensive and transparent LCA study comparing the life cycle environmental performance of conventional and electric vehicles. They found that "EVs powered by the present European electricity mix offer a 10–24 % decrease in global warming results relative to conventional diesel or gasoline vehicles assuming lifetimes of 150,000 km. However, EVs exhibit the potential for significant increases in human toxicity, freshwater eco-toxicity, freshwater eutrophication, and metal depletion impacts, largely emanating from the vehicle supply chain. Results are sensitive to assumptions regarding electricity source, use phase energy consumption, vehicle lifetime, and battery replacement schedules". For EVs production impacts thus become more important, while for conventional cars the use phase is by far the most important phase. The environmental performance of the EV can be improved by extending the lifetime of the EV, reducing the impacts of the EV production supply chain and by wider adoption of cleaner electricity sources.

Gregory et al. (2013) evaluated and compared the environmental impact (focusing on climate change) of five hand-drying systems: hands-under dryers, high-speed hands-under dryers, high-speed hands-in dryers, cotton roll towels and paper towels. They also developed a method for incorporating uncertainty in the comparison of these hand-drying systems to understand the statistical robustness of the difference between the environmental impacts of the five hand-drying systems. They conclude "with a high degree of confidence that the high-speed dryers have a lower impact than paper towels and cotton roll towels".

van der Giesen et al. (2014) recently published an LCA study focusing on global warming impacts on a subject that receives increasing interest as part of the discussion on the transition towards renewable energy sources: using CO_2 as a resource to produce sustainable liquid hydrocarbon fuels. Producing these fuels by only using solar energy labels them as solar fuels. Today, new technologies for energy production are often claimed to be "sustainable" before they have been developed properly and/or subjected to any kind of sustainability assessment. Therefore, van der Giesen et al. (2014) performed a quantitative LCA "to investigate some of the claims made in this discussion". They concluded "that producing liquid hydrocarbon fuels starting from CO_2 by using existing technologies requires much more energy than existing fuels. An improvement in life cycle CO_2 emissions is only found when solar energy and atmospheric CO_2 are used. Producing fuels from CO_2 is a very long-term niche at best, not the panacea suggested in the recent public discourse".

2.4 Developments in and Challenges to LCA

LCA is an active research area in terms of methodology as well as in practical aspects. Methodology developers are working hard on further improving different parts of the LCA method while also putting efforts in keeping the methods practically feasible. We here briefly discuss a subjective selection of ongoing developments and challenges that LCA is facing:

1. Development of consequential LCA
2. Broadening LCA to LCSA
3. Dealing with uncertainty in LCA
4. Streamlining LCA and LCA for ecodesign
5. Standardization of LCA methods beyond ISO

2.4.1 Development of Consequential LCA

Since the beginning of the present century, consequential LCA has significantly grown in terms of the number of case study applications (Zamagni et al. 2012; Guinée and Heijungs 2011). Consequential LCA is a modelling approach that aims to describe the consequences of a decision and often models various scenarios to examine possible consequences. Consequential LCAs include unit processes in the product system to the extent that they are expected to change as a consequence of a change in the demand for the product (Weidema et al. 2009). Attributional—as opposed to consequential—LCA aims to describe the environmentally relevant physical flows to and from a life cycle and its subsystems as they are, were or are expected to be. According to Schmidt (2010) and Weidema (2003), the core differences between CLCA and attributional LCA (ALCA) are that (1) consequential LCA includes the

suppliers actually affected by a change in demand instead of averages as in ALCA and (2) co-product allocation is avoided by system expansion instead of applying allocation factors. This change is modelled not over time but as a comparison of the situation with and without a specific demand; various product-related future scenarios are actually modelled. Future scenarios imply forecasting of future technologies, affluence and consumer behaviour and thus include large uncertainties, which increase as the term of the forecast becomes longer and the scope of the change becomes larger. Such uncertainties are inherent in modelling the future and thus also apply to an ALCA of a future system (Guinée and Heijungs 2011). However, due to the higher ambition of modelling how the world will look like, consequential LCA studies may have much bigger challenges in terms of data demand and models.

The modelling principles of ALCA and CLCA are the same; what distinguishes the two modes of LCA is the choice of the processes to be included in the system (Zamagni et al. 2012). Although the differences seem small, the application of the two modes to one case study (which can actually not be done, because they basically address different questions) may give significantly different results, not only between CLCA and ALCA but also between different (scenario) assumptions within CLCA (see Schmidt 2010). Bearing in mind that ALCA and CLCA can result in potentially significant differences, Plevin et al. (2014a) recently argued that "using ALCA to estimate climate-change mitigation benefits misleads policymakers" basically claiming that CLCA is conceptually superior to ALCA for supporting robust decision-making. Plevin et al. received a lot of criticism (Anex and Lifset 2014; Brandão et al. 2014; Dale and Kim 2014; Hertwich 2014; Plevin et al. 2014b; Suh and Yang 2014; Plevin et al. 2014c), but the debate has not been cleared yet, if ever.

2.4.2 Broadening LCA to Life Cycle Sustainability Assessment

As disclosed in Sect. 2.1.2, environmental life cycle assessment (LCA) has developed fast over the last three decades. LCA developed from merely energy analysis to a comprehensive environmental burden analysis in the 1970s; full-fledged life cycle impact assessment and life cycle costing (LCC) models were introduced in the 1980s and 1990s, and—amongst many other things—social LCA (S-LCA) and particularly consequential LCA gained ground in the first decade of the twenty-first century. With these latter developments, LCA broadened itself from a merely environmental LCA into a more comprehensive life cycle sustainability analysis (LCSA). Based on definitions by Klöpffer and Renner (2007; see also Klöpffer 2008) and Guinée et al. (2011), we can distinguish three dimensions along which LCSA expanded or may expand when compared to environmental LCA:

1. *Broadening* the scope of current LCA from mainly environmental *impacts* only to covering all three dimensions of sustainability (people, planet and prosperity): LCSA = LCA + LCC + S-LCA.

2. *Broadening* the scope from predominantly product-related *questions* (product level) to questions related to sector (sector level) or even economy-wide levels (economy level).
3. *Deepening* current LCA *methods* to also include other than just technological relations, e.g. physical relations (including limitations in available resources and land), economic and behavioural relations (including rebound effects), etc.

LCSA is still a rather new area of research and needs a lot of further research, particularly in designing practical methods. Nevertheless, several articles on LCSA have already been published in international journals, professional sections have been established such as the ISIE-LCSA section (http://www.is4ie.org/sections#LCSA), and LCSA has become a subject area in, for example, the *International Journal of LCA* (http://www.springer.com/environment/journal/11367).

2.4.3 Dealing with Uncertainty in LCA

A huge challenge for environmental LCA, and even huger for LCSA and CLCA, is how to deal with the many uncertainties related to LCA and LCA results. Until today, this issue was largely underexposed by methodology developers, practitioners, databases and software.

LCA is an integral method, bringing together data on many processes, data and models on impacts, assumptions on functional unit, system boundaries, allocation, etc. All of these data and (impact) models already bring many uncertainties with them, and in LCA we lump these uncertainties. LCA practitioners are usually already happy when they get the main part of the data they need. Getting a grip on the uncertainties related to these data and to the models they apply is generally a bridge too far, let alone to also quantify the influence on the results of different methodological choices and different assumptions. However, work on this is progressing as shown above in the chapter on Interpretation. Recently, Henriksson et al. (2015) even showed that statistically testing a hypothesis—requiring a predefined null hypothesis and quantification of uncertainties—is practically feasible (and, as a matter of fact, indispensable) for LCA-based product carbon footprints and provides more robust decision support. Main challenges now are to harmonize methods developed, to consistently elaborate and apply them over all sources of uncertainties throughout the whole LCA method and to collect the relevant input uncertainty data. Particularly the latter requires substantial efforts and can only be achieved by reporting LCA data comprehensively, transparently and in a publicly accessible way.

2.4.4 Simplifying and Streamlining LCA

Improving the LCA method is one thing, but keeping it practically feasible for users is another. Options for simplifying and streamlining LCA have been reviewed by Zamagni et al. (2008) and Pesonen and Horn (2013). They distinguished three

different types of simplified approaches: qualitative (e.g. qualitative matrices, checklists, expert panels), semi-quantitative (e.g. partly quantified matrices) and quantitative (simplified LCA). For all three categories practical methods have been developed. There is unfortunately no "one size fits all" solution for streamlining and simplification, and streamlined methods generally don't produce conclusions similar to those reached through full LCAs (Hunt et al. 1998). Zamagni et al. (2008) conclude that "the choice of the most suitable simplified method, or combination of simplified methods, depends on the type of results users are looking for". For a study "aiming at supporting a choice between several alternatives and a study aimed at identifying critical aspects and suggesting mitigation strategies different methods may be suitable. In the first case, it is much more important to have quantitative data than in the second case. In fact, in the first case the lack of a quantitative dimension would hinder the comparison and make it difficult to differentiate between products. On the other hand, problems could arise when in quantitative LCA, aspects that are difficult to quantify are handled qualitatively, because this qualitative information is often overlooked" (Hoschorner and Finnveden 2003; Zamagni et al. 2008). For this first type of question, the challenge is to simplified LCA models based on detailed LCA experiences for specific product (categories). The EU 5th framework project OMNIITOX developed models calculating characterization factors for assessing the potential toxic impacts of chemicals within the framework of LCA. The project developed two interrelated models in order to be able to provide LCA impact assessment characterization factors for toxic releases for as broad a range of chemicals as possible: (1) a base model representing a state-of-the-art multimedia model and (2) a simple model derived from the base model using statistical tools. Most simplified LCA models up till now have been developed in a rather "isolated" way, independent from detailed LCAs. The experiences in OMNIITOX (Birkved and Heijungs 2011) and reported by Huijbregts et al. (2006) seem useful to copy to simplified LCA models based on detailed LCAs for specific products. However, in some cases simplification may be much simpler than this. In the 1990s LCAs were performed on light bulbs, and the results were clearly dominated by the electricity needed for using the bulb. That steered all new bulb designs later on.

Simplification and streamlining are particularly relevant for ecodesigners. Ecodesign and LCA have always had a natural connection as they both strive to improve the environmental performance of product systems. Although the natural connection between the two seems obvious, there is also a natural tension between them. Whereas product designers in early stages often only have limited ideas—and thus data—on the eventual material and chemical composition of the product, an LCA study needs this information in order to produce useful supporting environmental information for the designer at stake. Designers' rules of thumb should be based on LCAs. The recent interest in meta-analysis of LCA (JIE 2012) is an interesting development in that respect. However, rules of thumb always have the danger of being wrong. In particular, they may turn out to be self-denying in the long run. For TV sets with a cathode ray tube, energy use was the most successful predictor

of overall life cycle impact. With the development of flat screens, the value of this rule of thumb has decreased dramatically. Now the presence of the rare metals may be a better predictor.

2.4.5 Standardization of LCA Methods Beyond ISO

As disclosed in Sect. 2.1.2, the past first decade of the twenty-first century showed basically divergence in LCA methods again. On top of this LCA started increasingly supporting policies and (bioenergy) performance-based regulations. For that LCA results needed to be robust and reliable. Both robustness and reliability can be challenged. For example, some of these performance-based regulations in the field of bioenergy adopted ALCA, others consequential LCA and others a hybrid of these approaches, resulting in different ratings and even altering preference orders for transportation fuels that were the subject of these regulations. In addition, most regulations lack reporting requirements on the uncertainty of the LCA results and subjective choices by the LCA practitioner, which can result in LCA results seeming more certain and scientifically objective than they really are. It became clear that ISO standards were not sufficient to tackle these problems, and thus product carbon footprint standards were developed all around the world, and for EU policy applications, the *International Reference Life Cycle Data System (ILCD) Handbook* was developed (ILCD 2010).

Standardizing and harmonizing methods for supporting policies and performance-based regulations is of utmost importance, but it will remain a challenge to properly balance this need with the equally justified need of science to progress and improve itself. It may thus make sense to distinguish between LCAs for science and LCAs for policy. "Whereas the former aims for completeness, the latter aims for robustness. The use of LCA in the policy context will benefit largely from the acceptance of this difference and by drawing up a guideline that is based on the aim of robustness" (Wardenaar et al. 2012) with an update every now and then to include progresses in science, the balance could be established. Eventually, also here there will be no "one size fits all" solution, and we will have to accept that different questions require different approaches.

A complication that is getting more and more pervasive is that software and databases de facto dictate the methodology. For instance, if scientists prove that one method for uncertainty calculations is superior to another but the available software continues to offer only the inferior method, practice will not change. Similarly, scientists may develop noise impact methods, but if unit process databases do not offer information on sound emissions, such developments will remain within the academic domain only.

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