Journal of Environmental Management 91 (2009) 1–21

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Review

Recent developments in Life Cycle Assessment

Göran Finnveden [a,*], Michael Z. Hauschild [b], Tomas Ekvall [c], Jeroen Guinée [d], Reinout Heijungs [d], Stefanie Hellweg [e], Annette Koehler [e], David Pennington [f], Sangwon Suh [g]

[a] *Division of Environmental Strategies Research – fms, Department of Urban Planning and Environment, School of Architecture and the Built Environment, Royal Institute of Technology (KTH), SE-100 44 Stockholm, Sweden*
[b] *Department of Management Engineering, Technical University of Denmark, Produktionstorvet 424, DK-2800 Lyngby, Denmark*
[c] *IVL Swedish Environmental Research Institute, PO Box 5302, SE-400 14 Göteborg, Sweden*
[d] *Institute of Environmental Sciences, Leiden University, PO Box 9518, 2300 RA, The Netherlands*
[e] *ETH Zurich, Institute of Environmental Engineering, 8093 Zurich, Switzerland*
[f] *European Commission, Directorate General Joint Research Centre (JRC), Institute of Environment and Sustainability (IES), Ispra, Italy*
[g] *Department of Bioproducts and Biosystems Engineering, College of Agriculture, Food and Natural Resource Sciences, 1390 Eckles Avenue, St. Paul, MN 55108, USA*

ARTICLE INFO

Article history:
Received 11 October 2008
Received in revised form
22 May 2009
Accepted 19 June 2009
Available online 29 August 2009

Keywords:
Life Cycle Assessment
LCA
Strategic Environmental Assessment
Risk assessment
LCC
Ecological footprint
Exergy analysis
Valuation
Weighting

ABSTRACT

Life Cycle Assessment is a tool to assess the environmental impacts and resources used throughout a product's life cycle, i.e., from raw material acquisition, via production and use phases, to waste management. The methodological development in LCA has been strong, and LCA is broadly applied in practice. The aim of this paper is to provide a review of recent developments of LCA methods. The focus is on some areas where there has been an intense methodological development during the last years. We also highlight some of the emerging issues. In relation to the Goal and Scope definition we especially discuss the distinction between attributional and consequential LCA. For the Inventory Analysis, this distinction is relevant when discussing system boundaries, data collection, and allocation. Also highlighted are developments concerning databases and Input–Output and hybrid LCA. In the sections on Life Cycle Impact Assessment we discuss the characteristics of the modelling as well as some recent developments for specific impact categories and weighting. In relation to the Interpretation the focus is on uncertainty analysis. Finally, we discuss recent developments in relation to some of the strengths and weaknesses of LCA.

1. Introduction

Climate change and other environmental threats have come more into focus during the last years. In order to meet these challenges, environmental considerations have to be integrated into a number of different types of decisions made both by business, individuals, and public administrations and policymakers (Nilsson and Eckerberg, 2007). Information on environmental aspects of different systems is thus needed, and many tools and indicators for assessing and benchmarking environmental impacts of different systems have been developed (e.g., Finnveden and Moberg, 2005; Ness et al., 2007). Examples include Life Cycle Assessment (LCA), Strategic Environmental Assessment (SEA), Environmental Impact Assessment (EIA), Environmental Risk Assessment (ERA), Cost-Benefit Analysis (CBA), Material Flow Analysis (MFA), and Ecological Footprint. In this paper, the emphasis is on LCA, but we will also address its influences from ERA, ecological footprint, etc. and vice-versa.

Life Cycle Assessment is a tool to assess the potential environmental impacts and resources used throughout a product's life-cycle, i.e., from raw material acquisition, via production and use phases, to waste management (ISO, 2006a). The waste management phase includes disposal as well as recycling. The term 'product' includes both goods and services (ISO, 2006a). LCA is a comprehensive assessment and considers all attributes or aspects of natural environment, human health, and resources (ISO, 2006a). The unique feature of LCA is the focus on products in a life-cycle perspective. The comprehensive scope of LCA is useful in order to avoid problem-shifting, for example, from one phase of the life-cycle to another, from one region to another, or from one environmental problem to another.

* Corresponding author. Tel.: +46 8 790 73 18.
E-mail addresses: goran.finnveden@infra.kth.se (G. Finnveden), mic@ipl.dtu.dk (M.Z. Hauschild), tomas.ekvall@ivl.se (T. Ekvall), guinee@cml.leidenuniv.nl (J. Guinée), heijungs@cml.leidenuniv.nl (R. Heijungs), stefanie.hellweg@ifu.baug.ethz.ch (S. Hellweg), annette.koehler@ifu.baug.ethz.ch (A. Koehler), david.pennington@jrc.it (D. Pennington), sangwon@umn.edu (S. Suh).

0301-4797/$ – see front matter © 2009 Elsevier Ltd. All rights reserved.
doi:10.1016/j.jenvman.2009.06.018

The interest in LCA grew rapidly during the 1990s, also when the first scientific publications emerged (e.g., Guinée et al., 1993a,b). At that time LCA was regarded with high expectations but its results were also often criticized (e.g., Udo de Haes, 1993; Ayres, 1995; Ehrenfeld, 1998; Krozer and Viz, 1998; Finnveden, 2000). Since then a strong development and harmonization has occurred resulting in an international standard (ISO, 2006a,b), complemented by a number of guidelines (e.g., Guinée et al., 2002) and textbooks (Wenzel et al., 1997; Baumann and Tillman, 2004). This has increased the maturity and methodological robustness of LCA. The method is still under development, however. There are also several ongoing international initiatives to help build consensus and provide recommendations, including the Life Cycle Initiative of the United Nations Environment Program (UNEP) and the Society of Environmental Toxicology and Chemistry (SETAC; UNEP, 2002), the European Platform for LCA of the European Commission (2008b), and the emerging International Reference Life Cycle Data System (ILCD).

Sustainability assessment of products or technologies is normally seen as encompassing impacts in three dimensions – the social, the environmental, and the economic (Elkington, 1998). On all the three a life cycle perspective is relevant to avoid problem shifting in the product system. With inspiration from environmental LCA, work has started on the development of methods for social LCA, and under the UNEP/SETAC Life Cycle Initiative, a project group is working on this topic (Grießhammer et al., 2006). Jørgensen and co-workers give a review of the state-of-the-art for social LCA (Jørgensen et al., 2008). Also Life Cycle Costing (LCC) is being developed as a method on its own. A recent overview was provided by SETAC (Hunkeler et al., 2008). The integration of the three sustainability dimensions is analysed and discussed in, for example, the EU 6th Framework Co-ordination Action for innovation in Life-Cycle Analysis for Sustainability (CALCAS, 2009) and the EU 7th Framework project Development and application of a standardized methodology for the PROspective SUstaInability assessment of TEchnologies (PROSUITE; Patel, 2009). This paper, however, focuses on environmental LCA.

There are four phases in an LCA study: Goal and Scope Definition, Life Cycle Inventory Analysis (LCI), Life Cycle Impact Assessment (LCIA), and Interpretation. The Goal and Scope Definition includes the reasons for carrying out the study, the intended application, and the intended audience (ISO, 2006a). It is also the place where the system boundaries of the study are described and the functional unit is defined. The functional unit is a quantitative measure of the functions that the goods (or service) provide. The result from the LCI is a compilation of the inputs (resources) and the outputs (emissions) from the product over its life-cycle in relation to the functional unit. The LCIA is aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts of the studied system (ISO, 2006a). In the Interpretation, the results from the previous phases are evaluated in relation to the goal and scope in order to reach conclusions and recommendations (ISO, 2006a).

The aim of this paper is to provide a review of recent developments in LCA methodology. We build upon previous reviews (e.g., Rebitzer et al., 2004; Pennington et al., 2004). Our focus is on areas with significant methodological development during the last years. We also highlight some of the emerging issues. In relation to the Goal and Scope Definition, we especially discuss the distinction between attributional and consequential LCA (Section 3). This distinction is relevant, for example, when defining system boundaries in the LCI which is further discussed in Section 5. The use of scenarios in LCA is discussed in Section 4. We also highlight developments concerning databases and input–output and hybrid LCA in Section 6. In the sections on LCIA (Sections 7 and 8) we discuss the characteristics of the modelling as well as some recent developments for specific impact categories (including spatial differentiation, toxicity, indoor air pollution, and impacts from use of land, resources, and water) and weighting. In relation to the interpretation we focus on uncertainty analysis (Section 9). Finally, we discuss these developments in relation to some of the strengths and weaknesses of LCA. First, however, some characteristics of LCA are elaborated in Section 2.

2. Some characteristics of Life Cycle Assessment vis-à-vis other environmental assessment tools

The focus on a product system in Life Cycle Assessment has some important implications for the nature of the impacts, which can be modelled in LCA:

- The product system is extended in time and space, and the emission inventory is often aggregated in a form which restricts knowledge about the geographical location of the individual emissions (this is further discussed in Section 8.3).
- The LCI results are also typically unaccompanied by information about the temporal course of the emission or the resulting concentrations in the receiving environment.
- The functional unit of the LCA refers to the assessment of an often rather small unit. The emissions to air, water, or soil in the inventory are determined as the functional unit's proportional share of the full emission from each process. The LCIA thus has to operate on mass loads representing a share (often nearly infinitesimal) of the full emission output from the processes.

The impacts, which can be calculated under such boundary conditions, thus represent the sum of impacts from emissions released years ago, from emissions released today and from emissions released some time in the future. Further, these impacts hit different ecosystems in different parts of the world. In the real world, environmental effects arise at a specific point in time and space and are often a function of the background pollution level. In LCA we have no knowledge about the simultaneous emissions from other processes outside the product system, which expose the same ecosystem/human cohorts, and no information about the background concentration of other substances in the system; LCA is thus no substitute for (Environmental) Risk Assessment. Instead, the results from the LCIA reflect the potential contributions to actual impacts or risks pending on the relevance and validity of the reference conditions assumed in the underlying models (Olsen et al., 2001; Hauschild, 2005; Tiruta-Barna et al., 2007).

LCA covers a diversity of environmental impacts and may include comparison across impact categories. It can therefore be argued that modelling should ideally be done with the same degree of realism for every impact to avoid introducing bias in the comparisons between categories. On this background, LCA aims for a comparable way of assessing impacts. This is often interpreted as aiming for best estimates in the modelling of all impacts. This can be in contrast to other tools, e.g., chemical risk assessment, where the first tiers will adopt a typically conservative approach applying realistic worst case estimates in order to be on the safe side in identifying situations posing potential risks of toxic effects above agreed thresholds (Hauschild and Pennington, 2002).

The quest for best estimates in modelling can bring LCIA into a potential conflict with the precautionary principle, which states that where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation (United Nations, 1992). In case of insufficient knowledge, the precautionary principle thus sanctions a conservative approach if the damage can be irreversible or serious in other ways. However, if best estimate modelling is used, risks for irreversible or other serious damages may be overlooked if, for example, they are not included in the best estimate models. This potential conflict with the precautionary principle can partially be overcome by adopting different social perspectives in LCIA (Hofstetter, 1998) and in the weighting step by assigning higher weights to those impacts where precautionary considerations justify it.

3. Attributional and consequential LCA

In the Goal and Scope Definition, questions or hypotheses should be formulated. This is an important phase since the appropriate LCA method depends on the purpose of the individual study (Consoli et al., 1993). Many attempts have been made to describe when different types of LCA are appropriate. We distinguish between two types of methods for LCA: attributional and consequential LCA. Attributional LCA is defined by its focus on describing the environmentally relevant physical flows to and from a life cycle and its subsystems. Consequential LCA is defined by its aim to describe how environmentally relevant flows will change in response to possible decisions (Curran et al., 2005). Similar distinctions have been made in several other publications (Ekvall, 1999), but often using other terms to denote the two types of LCA (such as descriptive versus change-oriented) and sometimes including further distinctions of subcategories within the two main types of LCA (Guinée et al., 2002).

Lundie et al. (2007a) argue that consequential LCA should be used for decision-making, but not when the difference between consequential and attributional LCA results is small, and not when the uncertainties in the consequential modelling outweigh the insights gained from it. When no decision is at hand, attributional LCA should be used because it is the more broadly applied method and because modelling consequences of decisions is somewhat pointless when no decision is at hand, according to Lundie et al. (2007a). Similar arguments have been presented by other authors (e.g., Tillman, 2000).

Weidema (2003) agrees that consequential LCA is more relevant for decision-making; however, he argues that it is also more relevant for increasing the understanding of the product chain and for identifying the processes and relations most important to improve. When a consequential LCA is applied for this purpose, Weidema (2003) argues that it generates a good basis for generating ideas on improvements.

Ekvall et al. (2005), on the other hand, argue that attributional and consequential LCA can both be used for decision-making and also for learning purposes. Consequential LCA is valid to assess environmental consequences of individual decisions or rules. Attributional LCA, on the other hand, is valid for the purpose of avoiding connections with systems with large environmental impacts. According to Ekvall et al. (2005) both of these purposes are legitimate.

As stated by several authors (e.g., Ekvall et al., 2005; Sandén and Karlström, 2007), attributional and consequential LCA can both be applied for modelling of future systems. Both can also be applied for modelling of past or current systems. Some recent case studies have been made using both attributional and consequential methods on the same product (Ekvall and Andrae, 2006; Thomassen et al., 2008) to illustrate the applicability of both approaches.

The different focuses of attributional and consequential LCA are reflected in several methodological choices in LCA (Tillman, 2000). One is the choice between average and marginal data in the modelling of subsystems of the life cycle. Average data for a system are those representing the average environmental burdens for producing a unit of the good and/or service in the system. Marginal data represent the effects of a small change in the output of goods and/or services from a system on the environmental burdens of the system. Attributional LCA excludes the use of marginal data. Instead, some sort of average data reflecting the actual physical flows should be used. On the other hand, in consequential LCAs marginal data are used when relevant for the purpose of assessing the consequences (Ekvall and Weidema, 2004). The decision to use marginal data can be significant for the modelling of electricity production, but also for land use (Kløverpris et al., 2008), pulpwood production (Åström, 2004), and many other products.

Different types of marginal effects can be included in a consequential LCA. Short-term effects are changes in the utilisation of the existing production capacity in existing production plants. Long-term effects involve changes in the production capacity and/or technology (Weidema et al., 1999). Energy systems analyses typically focus on the short-term effects. Weidema et al. (1999) suggest that the choice should be decided case by case, but states that the long-term effects are relevant in most cases. They present a five-step procedure to identify the long-term marginal technology.

Other authors argue that the most complete description of the consequences is obtained if short- and long-term effects are both accounted for (Eriksson et al., 2007). The combination of short- and long-term marginal effects in the Nordic electricity system has, for example, been investigated using a cost-optimizing dynamic model of this system (Mattsson et al., 2003). The results illustrate the potential complexity in the marginal effects and the large uncertainties involved in their identification. If a decision affects the timing of an investment in a power plant, it is also likely to affect the point in time where the power plant is taken out of use, the timing of the investment made to replace the first plant, and so on in a chain of cause-and-effect that do not seem to have an end. A decision can affect the production systems, at the margin, very far into the future. The uncertainty in the marginal effects grows with the time horizon. If an LCA includes marginal effects that occur far into the future, the uncertainty added can be larger than marginal effect itself.

The sphere of influence of a decision-maker includes not only the production of upstream products. It also includes, for example, the use of the upstream processes in other life cycles, the waste management of other products, and the level of economic activity in the society. Here we will briefly discuss such negative and positive feedback mechanisms, and also impacts on future resource uses.

A reduced use of a material, energy flow, etc. in a life cycle affects the balance between supply and demand for the good, which can increase the use of this good in other product systems. This is a negative feedback mechanism, since it offsets part of the initial change (Sandén and Karlström, 2007). Partial equilibrium analysis (Friedman, 1976; Weidema, 2003; Ekvall and Weidema, 2004; Lundie et al., 2007a) can quantify these effects in a consequential LCA, as demonstrated by Ekvall (2000), Ekvall and Andrae (2006), and Lesage et al. (2007a,b). Kløverpris et al. (2008) used a general equilibrium model to investigate how the global supply and use of arable land might be affected by a regional increase in cultivation.

If the efficiency measures are profitable, the savings in costs make it possible to increase the total economic activity. Such an increase results in a demand for the resource that – partly or completely – offsets the savings obtained through the original change. The increased economic activity is, of course, also likely to increase the demand for other resources. Valuable insights into such rebound effects can be generated through a general equilibrium model (Ibenholt, 2002).

Another rebound effect occurs when households gain money from choosing a cheaper product alternative. The consumption of a product is likely to increase when it grows cheaper, but part of the money saved is likely to be spent on other goods and services. Thiesen et al. (2008) used statistical data on private consumption in different income groups to estimate on what Danish households would spend the residual money in such a case. Weidema et al. (2008) used the environmental impacts of average consumption in EU-27 as a proxy for the impacts of marginal consumption in EU-27 countries.

Similar rebound effects can occur if the option chosen requires less time (in person-hours) or less space than the alternative options (Spielmann et al., 2008; Weidema et al., 2008).

If a decision is made to invest in an emerging technology, this contributes to the development of the technology and is likely to contribute to reducing the manufacturing cost of the equipment (Wright, 1936). Increased experience from manufacturing is also likely to improve the technological performance of the equipment (Claeson, 2000). The improved technology and reduced manufacturing costs both make the equipment easier to sell to other manufacturers. In this sense, the investment made by the first manufacturer makes it more likely that other manufacturers will make similar investments. This is a positive feedback mechanism. It might result in a snowball effect (Ekvall et al., 2006) and, eventually, in a radical change in the system. If the emerging technology has substantially lower environmental impacts than the traditional technologies, it can result in dramatically reduced environmental impacts (Sandén and Karlström, 2007). This positive feedback mechanism is apparently sometimes important to include in consequential LCAs. Sandén and Karlström (2007) did this by allocating the expected environmental gain of fuel cell buses to the learning investments in this technology. A more advance and, at least in theory, accurate method to model the consequences of an individual investment is to use systems models that include experience curves (Mattsson, 1997; Ekvall et al., 2006). There are also other positive feedback mechanisms discussed by Hertwich (2005a).

The current use of non-renewable resources may have the effect that future generations will have to use other resources with other environmental impacts (Stewart and Weidema, 2005). This has been the basis for several LCIA methods for resources (e.g., Steen, 1999; Goedkoop and Spriensma, 2000). However, as argued by Weidema et al. (2005), if current resource use leads to changes in the environmental interventions of future extractions, this should be modelled in the Inventory Analysis, at least in a consequential LCA, and not in the LCIA. If this is done, it will also lead to a discussion on the time issues in parallel to time in relation to landfilling discussed below (Section 5). It is, however, clear that the inclusion of future impacts from future resource uses as a consequence of current resource uses could have a significant impact on the results from the LCA.

The choice between attributional and consequential LCA will also influence system boundaries related to allocation (see Section 5) and can influence other methodological choices, such as the definition of functional unit (Rebitzer et al., 2004) and the choice of LCIA methods.

The environmental consequences of a decision apparently depend on a variety of environmental, technological, and economic mechanisms. Different concepts, approaches, and models have been developed to describe and analyse different mechanisms. No single person is an expert on all tools. For this reason, a comprehensive consequential LCA may require not only a combination of tools but also a combination of experts. The following procedure can be useful to deal with the scientific and administrative complexity involved (Ekvall et al., 2006):

1. make a preliminary list of the foreseeable types of consequences that are potentially important for the environment;
2. discuss which, if any, of the foreseeable consequences that should be quantified;
3. identify tools that are adequate to analyse and quantify each of these consequences;
4. create a network of experts on each of the tools that can participate in the consequential LCA;
5. analyse and describe the separate consequences; and
6. make a synthesis of the descriptions that describe the full quantifiable consequences of the decision.

The discussion in Step 2 should account for the costs as well as the benefits of quantifying each consequence. It would typically be based on qualitative judgement, but ideally on detailed cost-benefit analyses of the available methodological approaches. Such "method cost-benefit analyses" should be undertaken at a generic level but utilising experience from specific case studies as input (Lundie et al., 2007a).

A consequential LCA is likely to be conceptually complex, because it includes additional, economic concepts such as marginal production costs, elasticity of supply and demand, etc. Some of the models used in the analysis are also much less transparent than the linear and static model of a traditional LCA. Their results can also be very sensitive to assumptions, etc. (see, e.g., Mattsson et al., 2003). All these add to the risk that inadequate assumptions or other errors significantly affect the final LCA results. To reduce this risk, it is important to ensure that the various results regarding different consequences (Step 5) can be explained using credible arguments (again, see Mattsson et al., 2003).

The distinction between attributional and consequential LCA is one example of how choices in the Goal and Scope Definition of an LCA should influence methodological and data choices for the LCI and LCIA phases. There are also other choices which are of relevance. Guinée et al. (2002) make a similar distinction in types of LCA modelling but start by distinguishing three main types of questions, related to three main types of decision: occasional choices (concerning one-off fulfilment of a function), structural choices (concerning a function that is regularly supplied), and strategic choices (on how to supply a function for a long or even indefinite period of time). These different types of decisions may require different types of modelling (attributional or consequential) and different types of data, since they have different scales in terms of time and impacts.

4. Scenarios

In many applications, it is relevant to model future systems. This may, for example, be the case for consequential LCA where the impacts of a future possible decision are assessed, or in attributional LCAs aiming at assessing future technologies or systems. Whenever the systems that are modelled are future systems, a decision must be made on how to model the future. An easy way is of course to assume that the future is like the present and then model the present system. Sometimes this may be a good assumption. In other cases it may be more adequate to elaborate other future scenarios.

There are different types of scenarios (Weidema et al., 2004; Börjeson et al., 2006). Börjeson et al. (2006) suggest a typology based on the types of questions that are aimed at answering:

- Predictive scenarios aim at answering the question: What will happen?
- Explorative scenarios aim at answering the question: What can happen?
- Normative scenarios aim at answering the question: How can a specific target be reached?

Different techniques can be used to develop the different types of scenarios including workshops, time series modelling, and optimizing modelling resulting in both quantitative and qualitative scenarios (Börjeson et al., 2006).

All the different types of scenarios can be of interest in combination with LCA (Höjer et al., 2008). Predictive scenarios are useful, for example, for electricity production and other background processes in LCAs. If the time period is longer and the uncertainty in the forecasts increases, it may be useful to include several explorative scenarios, which together describe possible future developments of, for example, the energy systems (cf. Finnveden, 2008). This can be regarded as a special type of uncertainty analysis (cf. Section 9). Several studies on waste management systems have, for example, included explorative scenarios for the fuels that are competing with solid waste as a fuel, an aspect that may be decisive for the results of waste management LCAs (Björklund and Finnveden, 2005). Explorative scenarios have also been used in, for example, energy (Eriksson et al., 2007) and transportation studies (Spielmann et al., 2005). A normative scenario can be used in LCA as an explorative scenario describing what can happen. However, it may also be interesting to use LCA in normative scenario studies, for example, to evaluate environmental impacts in different backcasting scenarios (Höjer et al., 2008).

5. System boundaries and allocation

There are three major types of system boundaries in the LCI (Guinée et al., 2002):

- between the technical system and the environment,
- between significant and insignificant processes, and
- between the technological system under study and other technological systems.

Sometimes time and geographical limits are mentioned as system boundaries. However, these can be seen as special cases of boundaries towards the environment or towards other technological systems (see below).

In relation to the system boundary *between the technical system and the environment*, it can be noted that an LCA should cover the entire life cycle, although, e.g., cradle-to-gate studies can be called partial LCAs. Thus, the inputs should ideally be traced back to raw materials as found in nature. For example, crude oil can be an input to the life cycle, but not diesel oil since the latter is not found in nature but produced within the technical system. In parallel, the outputs should ideally be emissions to nature. Inputs to the system that have been drawn from the environment without previous human transformation and outputs released to the environment without subsequent human transformations are both called "elementary flows" in the ISO standard (ISO, 2006a).

In many cases, the system boundary between the technical system and the environment is obvious. However, when the life cycle includes forestry, agriculture (Audsley et al., 1994; Wegener Sleeswijk et al., 1996; Guinée et al., 2002), emissions to external wastewater systems, and landfills, the system boundary needs to be explicitly defined.

At landfills, the system boundary towards the environment can have a time dimension. It is generally accepted that the emissions from the landfill, in terms of gas and leachate, should be regarded as an output from the system, but not the waste itself (Finnveden et al., 1995). In practice this may sometimes be difficult due to lack of data on emissions from landfilling of the wastes. Furthermore, emissions from landfills can continue for very long time periods, thousands of years or longer, and the environmental impact of the pollutants might be different if they are emitted slowly. When modelling emissions from landfills different researchers have used different system boundaries in terms of time (e.g., Doka and Hischier, 2005; Obersteiner et al., 2007). Some have included emissions during 100 years or less based on legal responsibility limits, others during a hypothetical infinite time period until all the materials from the landfill have been emitted. The inclusion of long-term emissions and their weighting compared to short-term emissions can have an importance for the final results (e.g. Hellweg et al., 2003b). One solution to this problem can be to model both short-term emissions (for example, substances emitted during the first 100 years) and long-term emissions, and present the results separately (Finnveden et al., 1995). An alternative solution is to include an impact category called "stored toxicity" which keeps account of the amount of the toxic loads that are left in the waste at the end of the chosen time period (Christensen et al., 2007; Hauschild et al., 2008b). In both the cases, the problem of weighting impacts on different time-scales against each other remains.

The system boundary *between significant and insignificant processes* is difficult since it is generally not known in advance what data are insignificant. Moreover, once you know the data for a process, there is no specific reason to leave it out. A general approach can be to include easily accessible data, check the importance of the data, and refine if necessary and possible (Lindfors et al., 1995) performing the LCI and LCIA in iterative loops until the required precision has been achieved. The possibilities for doing this have increased with the development of better databases and the use of input–output analysis, both discussed below. Accumulated experience of, for example, the importance of capital goods (Frischknecht et al., 2007) also helps in separating significant from insignificant processes.

An LCA is often restricted to a product that is produced and/or used in a specific geographical area during a specific time period. It can also be limited to a specific production technology or to a level of technology (e.g., best available technology). The system boundary *towards other technological systems* also has to be defined, for example, when the LCA includes the so-called multi-functional processes. These occur when a process is shared between several product systems, and it is not clear to which product the environmental impacts should be allocated. There are three types of allocation problems: multi-output (in which a process produces several products; e.g., a refinery), multi-input (in which one process receives several waste products; e.g., a waste incinerator), and open-loop recycling (in which one waste product is recycled to another product; e.g. a used newspaper that is incinerated and the energy recovered as heat and electricity). Allocation is one of the most discussed methodological issues in LCA (e.g., Weidema, 2003; Ekvall and Finnveden, 2001; Curran, 2007; Heijungs and Guinée, 2007; Lundie et al., 2007a).

There are two principally different ways of handling multi-functional processes. One is to allocate (partition) the environmental impacts between the products. This can be done on the basis of several principles such as physical and chemical causation or economic value, or on an arbitrary choice of a physical parameter such as energy or mass. Both the allocations based on physical causation and on an arbitrary choice can result in an allocation based on a physical parameter. The difference lies in the reasons for choosing the parameter. Allocation based on physical causation is not always possible, since there is not always a physical causation involved. One example is the production of chlorine and sodium hydroxide which is done in an electrochemical process where the products are produced in fixed and stoichiometric amounts. In this case, the allocation cannot be based on physical or chemical causation because the ratio between produced chlorine and sodium hydroxide cannot be changed. Since allocation based on physical causation is not always possible, any claim that it is based on physical causation should be substantiated.

The other principle to approach the allocation problem is to avoid it by dividing the processes into subprocesses or expanding the system boundaries and include affected parts of other life cycles in the technological system under study (Tillman et al., 1994). For example, consider an LCA of waste management of newsprint in which incineration with energy recovery is compared to landfilling without gas extraction. In the approach with expanded system boundaries the emissions from the incineration are included in the product system, but an alternative competing source for energy is also included in the landfilling system so that both the systems fulfil the same functions (Tillman et al., 1994). In a variant of this, it is assumed that the energy from the newsprint incineration replaces energy from the competing source, which thus is avoided. The environmental impacts from the competing energy source are then subtracted from the environmental impacts from incineration of the waste paper. In this way the incineration system is credited for also producing heat and/or electricity. This form of resolving the multi-functionality problem is often referred to as the substitution by system expansion or avoided burden method.

The international standard (ISO, 2006b) gives some guidance on how to handle allocation problems. It states that whenever possible, subdivision or system expansion should be used to avoid allocation problems. If that is not possible, an allocation reflecting the physical (or chemical or biological) causations should be used if possible, and finally, if that is not feasible, allocation based on other measures, such as economic value (Guinée et al., 2004), mass, or energy, may be used. Lundie et al. (2007a) observe that the last option is still the most commonly applied. They also argue that system expansion does not eliminate allocation problems because new allocation problems are likely to occur when the system is expanded. Heijungs and Guinée (2007) argue that system expansion is impractical because of the large uncertainties involved and the lack of data on what is avoided. Weidema (2003), on the other hand, aims to demonstrate that system expansion is always possible. He also argues that the approach is likely to eliminate allocation problems because the new allocation problems that occur are likely to be less important than the original problems and can, eventually, be disregarded.

The ISO procedure has been criticized because it does not account for the dependency between the method and the goal of the LCA (e.g., Ekvall and Finnveden, 2001). The distinction between attributional and consequential LCAs above is, for example, significant in this context. The system boundaries in a consequential LCA are defined to include the activities contributing to the environmental consequences of a change, regardless of whether these are within or outside the cradle-to-grave system of the product investigated. Allocation problems may then often be avoided, as expanding the system boundaries to include affected processes outside the cradle-to-grave system is already an inherent part of a consequential LCA (Ekvall and Finnveden, 2001; Ekvall and Weidema, 2004). Weidema (2003) argues that all the allocation problems in consequential LCAs should be avoided through system expansion.

In contrast, for attributional LCA, allocation (partitioning) is often considered to be the correct method (Weidema, 2003; Lundie et al., 2007a; Thomassen et al., 2008). However, system expansion may also be used for attributional studies, if the concept of LCA is interpreted to be a tool not only for investigating individual life cycles but also for combinations of life cycles. To continue the example of waste management of newsprint given above, an attributional study may be performed using expanded system boundaries. In parallel to a consequential study, this would imply having two functional units in the expanded system (waste management and production of heat and/or electricity), but only one functional unit in the case of substitution by system expansion. The major difference between an attributional and a consequential study in this case is the type of data used. A consequential study would often use marginal data, whereas an attributional study typically would use average data.

With a more narrow interpretation of the LCA concept, where an LCA is a study of one product's life cycle only, it may be argued that a study that is using expanded system boundaries, giving it several functional units, should not be called an LCA at all. Instead such a study could be called an environmental systems analysis using LCA methodology.

The system boundary between the technological system under study and other technological systems is affected in various other ways by the choice of doing a consequential LCA (see Section 3).

6. Inventory database development and specific tools

6.1. Database development

An LCI requires a lot of data. Setting up inventory data can be one of the most labour- and time-intensive stages of an LCA. This is often challenging due to the lack of appropriate data for the product system under study (e.g., for chemicals production). In order to facilitate the LCI and avoid duplication in data compilation, many databases have therefore been developed in the last decades. These include public national or regional databases, industry databases, and consultants' databases that are often offered in combination with LCA software tools.

National or regional databases, which evolved from publicly funded projects, provide inventory data on a variety of products and basic services that are needed in every LCA, such as raw materials, electricity generation, transport processes, and waste services as well as sometimes complex products. Several national and international public databases have been released in the past, among them the Swedish SPINE@CPM database (CPM, 2007), the German PROBAS database (UBA, 2007), the Japanese JEMAI database (JEMAI, 2007; Narita et al., 2004), the US NREL database (NREL, 2004), the Australian LCI database (RMIT, 2007), the Swiss ecoinvent database (Ecoinvent, 2007), and the European Reference Life Cycle Database (ELCD) (European Commission, 2007a). Further databases are currently under development all over the world, for example, in Brazil, Canada, China, Germany, Malaysia, Thailand, and other countries.

Complementary to public LCA/LCI databases, and often a major source of their data, numerous international business associations worldwide have created their own inventory datasets as a proactive effort to support the demand for first-hand industry data. Among others, life cycle inventories are available from the aluminium (EAA, 2007), copper (Deutsches Kupferinstitut, 1995; Bruch et al., 1995), iron and steel (IISI, 2007), plastics (APME, 2007), and paper and board (FEFCO, 2006) industries, covering a wide range of economic activities from extraction of, for example, metal resources to the manufacturing of combinations of materials such as metals alloys and currugated board.

Some databases, such as the ecoinvent and the US NREL databases, provide also data modules used to build inventories on a disaggregated unit-process level (e.g., for a chemical processing facility with multiple products such as a refinery). This means that the inputs and outputs are recorded per production step, in addition to aggregated data sets (e.g., cradle-to-gate). In contrast, many other databases, such as most of the databases provided by industry associations, supply inventory data as already-aggregated results (such as cradle-to-gate sub-systems), which specify the elementary flows (resource expenditures, emissions, and wastes) aggregated for all processes involved, for example, per mass unit of product manufactured.

Both aggregated and unit-process data sets are useful for modelling processes in LCA. Aggregated data can be, in particular with regard to industry data, more readily available, as company confidentiality may be guaranteed. Such data are often used as background data for modelling the production of, e.g., aluminium, steel, electricity, etc., where the practitioner typically does not know the exact source of the material or energy except possibly the region or market. This is particularly the case for globally traded products. The usefulness of aggregated industry data is sometimes discussed, however. Whereas some argue that such data are more reliable and representative, others would caution against hidden biases and missing transparency, suggesting the importance of ensuring the quality of such data.

Unit process data, in contrast to average data, often refer to specific technologies. This provides the possibility for tailored inventories, choosing the technologies that are in place in the case investigated, and allowing the study to focus on, for example, best available technologies and different mixes. This is particularly useful when a specific chain of processes is being considered and for foreground data where there is a good knowledge of what technologies are used. Furthermore, unit process data allow the LCA practitioner to review underlying details of the process data and methodological choices, make changes in an inventory such as for the energy mix used, and sometimes even to choose another allocation principle.

The majority of database systems are based on average data representing average production and supply conditions for goods and services, and thus employ the attributional modelling approach. Quality and consistency are key issues related to inventory data. While within specific databases, these are ensured to some extent, across databases there can exist significant differences. This includes data documentation (e.g., different data formats), modelling approaches (e.g., consideration of capital goods, allocation procedures), and nomenclature of flows and environmental exchanges denoted in the inventories. Bridging these differences to ensure the efficient exchange of data is one of the challenges in the field of LCA.

The UNEP/SETAC Life Cycle Initiative (UNEP, 2002) and the International Reference Life Cycle Data System (ILCD) (European Commission, 2008b) are trying to address inventory data, among other things, in the contexts of consistency and quality assurance building on existing examples and achievements, e.g., the ISO TC 14048. Another and complementary approach to help support the exchange of data amongst the many LCA tools and databases in current practice is a format conversion tool, to convert LCA data formats from one to another (e.g., Ciroth, 2007; Format Converter, 2008).

6.2. Tools for modelling subsystems and unit processes

There are a number of commercially available softwares for LCA. A registry of LCA tools (including software) and database providers is available from the EC.[1] LCA software typically includes some of the databases mentioned above as well as others. Yet, more specific processes, such as the production of fine chemicals or the treatment of a variety of complex waste and wastewater flows, are usually beyond the scope of inventory databases. In such cases, detailed models with enhanced flexibility may be useful. They can address the particular needs of different users and allow for the calculation of inventory data with reasonable effort. Some examples of such tools are described below.

[1] http://lca.jrc.ec.europa.eu/

The most prominent and traditional examples are tools for waste-management modelling. For instance, for municipal solid waste treatment various such inventory tools are available for direct sanitary landfills (Nielsen and Hauschild, 1998; Doka, 2003), mechanical biological treatment (Hellweg et al., 2003a), as well as for various incineration technologies (Sundqvist et al., 1997; Kremer et al., 1998; Hellweg et al., 2001; Doka, 2003). For integrated waste management a number of dedicated tools have been developed (e.g. McDougall et al., 2001; Eriksson et al., 2002; Solano et al., 2002; Christensen et al., 2007; den Boer et al., 2007; UK Environment Agency, 2008) and these typically include models for a number of different waste treatment methods. They enable the quantification of emissions, use of ancillaries, and energy production/use as a function of waste composition and technologies applied. The incineration and co-processing of industrial wastes in clinker kilns have also been modelled in a similar fashion (Seyler et al., 2005a,b). Cement manufacture was also modelled in a life-cycle approach by Gäbel et al. (2004). Tools have long been available for modelling municipal wastewater treatment (Zimmermann et al., 1996; Doka, 2003) and lately also for industrial wastewater systems (Köhler et al., 2007). In all these cases, causal relationships between ancillary use or emissions and waste(water) elemental input composition have been set up on the basis of measurements (e.g., concentration measurements in the flue gas and analysis of waste input), assuming mostly linear relationships.

Tools have also been developed for chemicals production and recycling technologies (Jimenez-Gonzales et al., 2000). Since direct input/output relationships could not be based on measurements in these cases, expert judgments were used (Geisler et al., 2004) or a statistical evaluation of empirical observations was performed (Capello et al., 2005). Another innovative approach is to use neural networks as a black-box model to estimate inventory data of fine chemical production (Wernet et al., 2008). These models are helpful because the producing companies tend to maintain LCI data of chemicals confidential. Also, there are more than 100,000 chemicals on the market and it would be impossible to do a detailed LCA on all of them. Therefore, these tools help abridging significant data gaps in LCA. For the estimation of pesticide emissions from agricultural field applications, an inventory tool has been developed taking into account the properties (physico-chemical and biological) of the pesticide ingredients and the conditions under which it is applied to the field (Birkved and Hauschild, 2006).

6.3. Input–output and hybrid LCA

Input–Output Analysis (IOA) is a field of economics that deals with the connections between industry sectors and households in a national economy in the form of supply and consumption of goods and services, formation of capital, and exchange of income and labour. Much of its basic framework was established in the first half of the 20th century with the life-long devotion of Wassily Leontief, for which he received the Nobel Prize in 1973. Recognizing its usefulness, national statistical authorities started to compile Input–Output Tables (IOTs) as a part of national accounts around the 1950s, and nowadays most industrialized nations regularly publish their IOTs. The IOT states, in average monetary terms and for each economic sector, how much the sector buys from each of the other sectors, for each unit produced in the sector.

An IOT becomes a powerful tool for LCA practitioners, when information on average resources use and environmental emissions from each sector are added to the table. Then, it can be used to estimate the environmental interventions generated throughout the upstream supply-chain to deliver a certain amount of different goods and services.

The calculations are based on data for industrial sectors, and will thus provide results for the "average product" from the sector. These data are used as approximations for specific products or product groups from the sector. The precision of this approximation can be poor and depends on how typical or atypical the studied product or product group is in relation to the other products from this sector. It is clear that the more atypical the product is in relation to the sector in question, the less adequate can be the approximation.

Lave et al. (1995) argued that conventional process-LCAs may miss significant parts of environmental interventions in their LCI, particularly for products where environmental emissions occur mostly in far upstream processes (Lenzen, 2000). This is because process-LCAs cannot include all processes. Ideally, those that are left out should have an insignificant contribution to the results, but this may not always be true. For many practitioners, however, Input–Output (IO)-LCA is not an attractive alternative to process-LCA for detailed product-level LCA, as its sector resolution is much too coarse for major LCA applications such as raw materials selection and process redesign.

What emerged was a hybrid technique combining the advantages of both process-LCA and IO-LCA (see Suh et al., 2004). Although Moriguchi et al. (1993) pioneered the usefulness of the hybrid technique, it was not until the late 1990s and early 2000 that hybrid LCA became widely acknowledged by LCA practitioners. Moriguchi et al. (1993) combined IOA and process analysis by analysing the main processes in detail using process-LCA, while estimating far upstream flows remotely connected to the main processes using IOA (see also Treloar et al., 2000; Suh and Huppes, 2002). This type of combination is called the tiered-hybrid approach. Joshi (1999) disaggregated a sector of an IOT into products to improve sector resolution for detailed applications forming another type of combination called the IO-based hybrid approach. Suh (2004) integrated the matrix approaches of LCI (see, e.g., Heijungs, 1994; Heijungs and Suh, 2002) and IOA in a consistent computational framework forming the integrated hybrid approach (see Suh, 2006; Peters and Hertwich, 2006; Kondo and Nakamura, 2004). Suh and Huppes (2005) provide a review of LCI approaches including hybrid approaches and their advantages and disadvantages.

In relation to the discussion on system boundaries in Section 5, the use of IOA through hybrid technique can help provide a more complete picture. The need to identify and exclude insignificant processes is reduced or eliminated, because the IOT accounts for all upstream processes. IOA is in these cases primarily used, not to do the LCA, but to estimate LCA data. The Missing Inventory Estimation Tool (MIET, Suh and Huppes, 2002) is an example of this development.

With regard to the discussion on attributional and consequential LCA, it can be noted that the average data contained in an IOA are adequate for attributional LCA but less so for consequential LCA. They typically do not describe how the resource uses and emissions of a sector are affected by possible decisions.

While stand-alone applications of IO-LCAs may fall short in providing information at process-level detail, its encompassing nature has been applied to applications for environmental policy supports at a macro-level (Hertwich, 2005b; Huppes et al., 2006; Tukker and Jansen, 2006). As a part of the activities of the European Commission's Integrated Product Policy (IPP) Communication, for example, there was a need to identify key products and services produced and consumed within the European Union countries that impose significant environmental loads. The IO-LCA was recognized as an appropriate tool for identifying such hotspots at the EU and national-scales in these IPP projects (Huppes et al., 2006; Palm et al., 2006; Tukker and Jansen, 2006; Weidema et al., 2006). Indicators for IPP have also been suggested based on IO-LCA (Palm et al., 2006; Björklund et al., 2007).

Up-to-date and comprehensive IO databases with environmental extensions are essential for applying IO and hybrid techniques for LCA. The question whether available databases are robust enough has been raised. Normally the used IOA data come from national statistics as a part of environmental and economic accounts developed within the statistical agencies, and are thus as accurate as they are. Progresses in environmental IO database development for LCA have been and are being made in various countries.

The most easily accessible database is perhaps the EIO-LCA database by Carnegie Mellon University, which provides on-line results for a number of environmental interventions (GDI, 2007). Another on-line tool is available from Statistics Sweden (SCB, 2008). The Comprehensive Environmental Data Archive (CEDA) 3.0 database is a peer reviewed IO-LCA database that combines the detailed US IOT and various data sources on environmental emissions and resource use encompassing 1344 environmental intervention items (Suh, 2005). It has been adapted to various policy applications and analytical tools, assessment of sustainable investment portfolios, and carbon footprint calculators (Huppes et al., 2006; Koellner et al., 2007). The 3EID is a peer-reviewed database for Japan that connects energy use and air pollutants emission data to Japanese IOT (Nansai et al., 2002, 2003). Besides, large-scale environmental IO database projects for LCA purposes are currently in progress in other regions. Particular attention is given to international trade and associated environmental impacts in these new initiatives, notably in the ongoing EXIOPOL project. We refer to Suh et al. (2004) for a more detailed survey of IO-LCA databases.

7. Impact assessment; general structure

The purpose of the Life Cycle Impact Assessment (LCIA) is to provide additional information to help assess the results from the Inventory Analysis so as to better understand their environmental significance (ISO, 2006a). Thus, the LCIA should interpret the inventory results into their potential impacts on what is referred to as the "areas of protection" of the LCIA (Consoli et al., 1993), i.e., the entities that we want to protect by using the LCA. Today, there is acceptance in the LCA community that the protection areas of Life Cycle Assessment are human health, natural environment, natural resources, and to some extent man-made environment (Udo de Haes et al., 1999, 2002).

Impacts on the areas of protection are modelled applying the best available knowledge about relationships between interventions in the form of resource extractions, emissions, land and water use, and their impacts in the environment, as illustrated in Fig. 1 for emissions of substances.

For greenhouse gases like CO_2 and CH_4, one of the first impacts after emission would be the increment they cause in the atmosphere's ability to absorb infrared radiation. This impact leads to other impacts among which are an increase in the atmospheric heat content and temperature, propagating to the global marine and soil compartments causing changes in regional and global climates and sea-level rise, eventually leading to damage to several of the areas of protection: human health, natural environment, and man-made resources. The fate processes would be the degradation and transport of the gas in the troposphere, the stratosphere, and the global water and soil compartments and they would be continuing along the chain of impacts all the way from emission to the areas of protection.

In Fig. 1 a distinction is made between midpoint and endpoint, where endpoint indicators are defined at the level of the areas of protection and midpoint indicators indicate impacts somewhere between the emission and the endpoint.

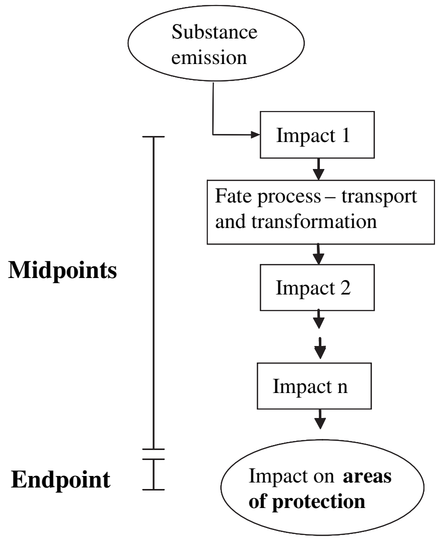

Fig. 1. Schematic presentation of an environmental mechanism underlying the modelling of impacts and damages in Life Cycle Impact Assessment (adapted from Hauschild and Potting, 2005).

Taking the areas of protection as starting point, LCIA applies a holistic perspective on environmental impacts and in principle attempts to model any impact from the product system which can be expected to damage one or more areas of protection. This means that LCIA addresses not only the toxic impacts (as chemical risk assessment does) but also the other impacts associated with emissions of air pollutants (global warming, stratospheric ozone depletion, acidification, photochemical ozone, and smog formation) or waterborne discharges (eutrophication and oxygen depletion) as well as the environmental impacts from different forms of land and water use, from noise and from radiation, and the use and loss of renewable and non-renewable resources.

In general, ISO restricts LCIA to environmental impacts and does not address the other two dimensions of sustainability – the social impacts and the economic aspects in the life cycle (Hunkeler and Rebitzer, 2005). The omission of social impacts from LCIA is to some degree inconsistent with the defined areas of protection, since social impacts will often lead to impacts on at least one of these areas, human health, and, as mentioned above, attempts are on-going to develop LCIA for social impacts, e.g., under the UNEP/SETAC Life Cycle Initiative (Grießhammer et al., 2006; Jørgensen et al., 2008).

According to the ISO standard on LCA (ISO, 2006a,b), LCIA involves:

Selection of impact categories and classification. Involves identification of the categories of environmental impacts which is of relevance to the study. This has normally been done in a general manner outside the study, which means that existing impact categories can just be adopted. The classification assigns the emissions from the inventory to these impact categories according to the substances' ability to contribute to different environmental problems.

Selection of characterisation methods and characterisation. The relevant characterisation models are selected and the impact of each emission is modelled quantitatively according to the environmental mechanism (Fig. 1) and expressed as an impact score in a unit common to all contributions within the impact category (e.g., kg CO_2-equivalents for greenhouse gases contributing to the impact category climate change) applying the concept of characterisation factors which for each substance expresses its potential impact in terms of the common unit of the category indicator. For example, for climate change an often used characterisation factor is the global warming potential for time horizon 100 years (GWP_{100}). Characterisation allows summing the contributions from all emissions and resource extractions within each impact category, translating the inventory data into a profile of environmental impact scores.

Normalisation where the results from the characterisation are related to reference values. Normalisation expresses the relative magnitude of the impact scores on a scale which is common to all the categories of impact (typically the background impact from society's total activities) in order to facilitate the interpretation of the results.

The final steps of the Impact Assessment include *Grouping* or *Weighting* of the different environmental impact categories and resource consumptions reflecting the relative importance they are assigned in the study. Weighting may be needed when trade-off situations occur, e.g., where improvements in one impact score are obtained at the expense of another impact score. Where normalisation expresses the relative magnitudes of the impact scores and resource consumptions, weighting expresses their relative *significance* in accordance with the goal of the study.

According to the ISO standard on LCA, selection of impact categories, classification, and characterisation are mandatory steps in LCIA, while normalisation and weighting are optional (ISO, 2006b). The weighting step is the most normative part of the method with its application of preferences and stakeholder values in a ranking, grouping or quantitative weighting of the impact categories. There is no objective way to perform weighting and hence, no "correct" set of ranks or weighting factors. The ISO standard for LCIA does not permit weighting to be performed in studies supporting comparative assertions disclosed to the public (ISO, 2006b).

While the ISO standard for LCIA (ISO, 2006b) presents the framework and some general principles to adhere to, it refrains from a standardisation of more detailed methodological choices. Over the last decade, several well-documented LCIA methods have been developed filling this gap (e.g., Heijungs et al., 1992; Wenzel et al., 1997; Hauschild and Wenzel, 1998; Goedkoop and Spriensma, 2000; Steen, 1999; Guinée et al., 2002; Bare et al., 2003; Itsubo and Inaba, 2003; Jolliet et al., 2003a).

8. Current developments in Life Cycle Impact Assessment

8.1. Towards a recommended practice for LCIA

Life Cycle Assessment is still a young discipline, mainly developed from the mid-1980s until now. Throughout the 1990s, a number of consecutive international working groups in SETAC took the method development and consensus building a good step forwards (Consoli et al., 1993; Fava et al., 1993; Udo de Haes, 1996; Udo de Haes et al., 2002), but LCIA is still a discipline in vivid development.

Today, several LCIA methods are available, and there is not always an obvious choice between them. In spite of resemblance between some of them, there can be important differences in their results, not least for the toxic impacts – differences which can lead to conclusions which depend on choice of LCIA method (Dreyer et al., 2003).

While standard bodies such as ISO have generally refrained from standardisation of the more detailed methodological choices, there are now international activities which aim at providing such recommendations. An element within the UNEP/SETAC Life Cycled Initiative (UNEP, 2002) is to help identify best practice for Life Cycle Assessment within the framework laid out by the ISO standards and to make data and methods for performing LCA available and applicable worldwide (UNEP, 2002). Recommendations are based on working groups focusing on the scientific validity of the methods and their feasibility in LCIA.

Building on recommendations from the UNEP/SETAC's scientific working groups as an important starting point and working together with UNEP, the European Commission in consultation with several non-EU countries, industry associations, as well as scientific experts is further facilitating the development of formal international recommendations for LCIA through the International Reference Life Cycle Data System (ILCD) (European Commission, 2008b). These recommendations will include characterisation models and operational characterisation factors for important substances. They address the midpoint level as well as the endpoint level.

8.2. Midpoints and endpoints in characterisation modelling

Traditional characterisation methods are examples of midpoint modelling, meaning that they choose an indicator somewhere between emission and endpoint in the environmental mechanism (a "midpoint") and model the impact on this indicator. The indicator is typically chosen where it is judged that further modelling is not feasible or involves too large uncertainties, or where a relative comparison can be made without the need for further modelling. Representatives of this midpoint school are Heijungs et al. (1992), Guinée et al. (2002), Wenzel et al. (1997), Hauschild and Potting (2005), and Bare et al. (2003).

An alternative school of characterisation modelling takes as a starting point that the purpose of LCA is to reveal contributions to impacts on the areas of protection and that consequently, LCIA must model the impacts on these. In this approach, characterisation modelling must include the entire environmental mechanism, since the areas of protection are located at the endpoint of it (see Fig. 1). Consequently, this approach to characterisation modelling is referred to as *endpoint modelling*. Examples of this endpoint school are methodologies developed by Goedkoop and Spriensma (2000), Steen (1999), and Itsubo and Inaba (2003).

Both approaches can include partial modelling, mid-point approaches because the whole environmental mechanism until the end-point is not included, and end-point approaches because typically some end-points are included but not all. Proponents of the endpoint school further point out that in case weighting is needed, it is limited to weighting or ranking of the areas of protection if an end-point approach is chosen. At the same time, it is necessary to weight the midpoint-based impact scores in order to compare across impact categories even within areas of protection (Bare et al., 2000). For both approaches the weighting is often based on social science and external costing methods discussed below. This weighting may, however, for the mid-point approaches, qualitatively involve the science-based analysis of the unmodelled parts of the environmental mechanism, considering aspects like severity and reversibility of the impacts on endpoints, their geographical extent and expected duration, and the uncertainty of the models predicting them (Hauschild and Wenzel, 1998) – in other words aspects which the endpoint approaches try to model quantitatively. But, obviously, the weighting of midpoint results introduces additional uncertainties to the midpoint approaches if it is a weighting on endpoints that is wanted. Different types of uncertainties thus have to be pondered when choosing the position of the midpoint impact indicator; the statistical uncertainty of the models and parameters which are used for modelling the indicator and the uncertainty of interpreting the resulting indicators in terms of damage to the areas of protection. It may also be argued that an advantage of midpoint approaches is that they make it possible to give consideration to the precautionary principle and give an extra weight to uncertain and partly unknown aspects, one example may be climate change. This is in contrast to endpoint modelling which typically will give no consideration to impacts which cannot be modelled and thus are more uncertain and unknown. A potential benefit of the end-point methods is the increased possibilities of comparing impacts at an end-point level, e.g., human health impacts from toxicological effects from those of climate change, if both can be modelled within the same framework.

The two schools are not necessarily incompatible. In fact, workshops focused on this topic concluded that a single framework with both should be used (Bare et al., 2000). In principle they agree in the quest of modelling relevant impact indicators. The principal discrepancy lies in the evaluation of whether the uncertainty in midpoint versus endpoint modelling is justified by the improved interpretation of the results, and the answer varies between the different categories of impact and different practitioners/clients. While reliable endpoint modelling seems within reach for some of the impact categories like acidification, cancer effects, and photochemical ozone formation, it is still developing for climate change, where a midpoint approach will choose the indicator rather early in the environmental mechanism (at the level of radiative forcing), and where endpoint modelling is encumbered with large uncertainties due to the many unknowns of the global climate system and to the long time horizon of some of the involved balances. However, there is often also a practical discrepancy when the underlying models of midpoint and endpoint approaches differ in the systems they model. Such discrepancies are confusing, and often unnecessary. It is therefore desirable that methods for LCA should be harmonized in some of the detailed choices while allowing a certain degree in freedom as to the main principles, in the current case their orientation towards midpoint or endpoint indicators. Harmonization of these underlying models has been done in, e.g., the ReCiPe project (Heijungs et al., 2003), IMPACT2002+ (Jolliet et al., 2003a), and the Life Cycle Impact Assessment Method based on endpoint modelling (LIME2) which is being developed as part of the second LCA National Project of Japan.

8.3. Spatial differentiation – getting the exposure right

The impacts caused by an emission depend on

- the quantity of substance emitted
- the properties of the substance
- the characteristics of the emitting source
- the receiving environment

The site-generic approach (or global default) followed in current characterisation modelling only includes the first two aspects, inherently assuming a global set of average/standard conditions as regards the properties of the source and the receiving environment. For truly global impact categories like climate change and stratospheric ozone depletion, this is not a problem since the impact is independent of where the emission occurs. For the other impacts modelled in LCIA, however, the situation can be different. They are often regional or even local in nature, and a global set of standard conditions can disregard large and unknown variations in the actual exposure of the sensitive parts of the environment. Sometimes differences in sensitivities of the receiving environment can have a stronger influence on the resulting impact than differences in inherent properties of the substance that contribute to the impact (Potting and Hauschild, 1997; Bare et al., 2003). At the same time, spatial differences can be reduced in the case of sources from multiple locations, particularly when these result in uniform emission distributions.

The modelled impacts in LCIA must show accordance with the actual impacts associated with the product system, if the decisions based on the LCA shall lead to environmental improvements. Therefore, spatial differentiation can be relevant in LCIA (Udo de Haes et al., 1999). It is equally acknowledged that this will increase the complexity of LCA, requiring more information in some cases about emissions and more differentiation in the impact assessment. Even though the specific location and properties of the receiving environment will only be known for very few processes, the country and possibly the region will, however, sometimes be known for many processes in an LCA, where it is needed for modelling the transportation activities in the life cycle. It is thus possible, at least, to base an exposure modelling on this type of geographical information and derive site-dependent characterisation factors which depend on the country or region of emission as well as on the properties of the substance.

Several groups have worked on developing site-dependent characterisation for LCIA (Krewitt et al., 1998; Huijbregts et al., 2000a), and recently, methods supporting site-dependent characterisation of a range of non-global impact categories was published for processes in Europe (Hauschild and Potting, 2005; Hettelingh et al., 2005; Seppälä et al., 2006) and in the US (Bare et al., 2003). There are some differences between these data sets partly related to the different definitions of the characterisation factors (Seppälä et al., 2006). There has also been a discussion on the needs of defining site-dependent characterisation factors within countries (Finnveden and Nilsson, 2005). Another type of site-dependent aspect that may need consideration is the difference between emissions from low and high stacks which may be of importance especially for impacts on human health (Finnveden and Nilsson, 2005; van Zelm et al., 2008). Modelling country-dependent characterisation factors, Potting and co-workers find that the variation in acidification impact can be as high as three orders of magnitude between different countries within Europe (Potting et al., 1998).

Whether site-dependent factors reduce uncertainty compared to using generic defaults depends on the impact category of concern and it may also depend on the case. It is therefore essential for the research community to identify under what conditions uncertainties can be reduced in LCAs to justify not using generic inventory and impact assessment data.

8.4. *Resources*

For resources, a distinction is generally made between biotic and abiotic resources. Biotic resources are considered important but have not yet received much attention.

Abiotic resource depletion is one of the most frequently discussed impact categories and there are consequently a wide variety of methods available for characterising contributions to this category (Pennington et al., 2004). To a large extent these different methods reflect differences in problem definition, and earlier reviews have concluded that it is difficult to choose between the different approaches (Lindeijer et al., 2002). However, since then some new developments can influence the debate. One is the insight discussed in Section 3 that methods based on environmental impacts from future extractions should be included in the Inventory Analysis and not in the Impact Assessment (Weidema et al., 2005). This means that several of the often used LCIA methods for abiotic resources (e.g., Ecoindicator 99, EPS method, and Lime and Impact 2002+) are debatable (Finnveden, 2005). This means that there are essentially two groups of methods available for characterisation of abiotic resources:

- Methods related to some measure of available resources or reserves and extraction rates. Different approaches exist based on different measures of the reserves, e.g., technically and economically available reserves (Wenzel et al., 1997) or ultimately available reserves (Guinée et al., 2002) and extraction rates.
- Methods based on exergy consumption or entropy production. Exergy is a measure of available energy and entropy can be interpreted as a measure of disorder. Methods and data based on this approach have been developed for LCA by Finnveden and Östlund (1997) and data for a large number of resources were recently published by Bösch et al. (2007). This is interesting since one of the arguments against a method based on exergy consumption has been the lack of data which now may be largely removed. Dewulf et al. (2007) suggested that exergy data on fossils, nuclear and metal ores, minerals, air, water, land occupation, and renewable energy sources can be calculated and aggregated as the cumulative exergy extraction from the natural environment.

8.5. *Impacts of land use*

Land use is an elementary flow that leads to an impact category, or a group of impact categories, that have been discussed quite a lot (e.g., Lindeijer et al., 2002; Mila i Canals et al., 2007a,b; Udo de Haes, 2006). Both the land occupation and the land transformation involved in agriculture and forestry, but also other activities such as mining and transportation can have significant impacts, both positive and negative. However, there is currently no agreement on how these impacts should be included in an LCA. Land use will affect three of the areas of protection directly: natural environment, natural resources and manmade environment, and human health indirectly. Examples of impacts include loss of biodiversity, loss of soil quality, and loss of biotic production potential (Mila i Canals et al., 2007a) but the list of potential impacts to include is longer (Udo de Haes, 2006). Since there are several types of impacts, it may be necessary to have several impact categories.

Several methods have been suggested in the literature on how to include land use impacts (see, e.g., reviews in Lindeijer et al. (2002), Mila i Canals et al. (2007a), and Pennington et al. (2004) and recent publications by Koellner and Scholz (2007, 2008) and Michelsen (2008)). The approach based on cumulatative exergy extraction mentioned above and the ecological footprint discussed below are also approaches which include some aspects of land use. There are, however, limited experiences of using and comparing the different methods in practice. Their general applicability is therefore generally untested. A working group within the UNEP/SETAC Life Cycle Initiative has been formed and will address these issues further.

8.6. *Impact from water use*

Freshwater as a resource provides fundamental functions for humans and the environment, and is thus relevant for all four areas of protection. In its first operating phase, the UNEP/SETAC Life Cycle Initiative recognized the high global and regional significance of freshwater resources and their limited availability on a global level (Stewart and Goedkoop, 2003) and clearly expressed the need for an assessment of water resource consumption (Jolliet, 2003) which is continuously growing due to economic, demographic, and climate change influences.

At the level of life cycle inventories, water flows are sometimes reported as input parameters, but only little attention is given to the differentiation of water types and water quality criteria which may be important for the subsequent Impact Assessment. Particularly industrial water use is hardly documented. To date no generally accepted standards for water-use reporting in LCA exist (Koehler, 2008). Water quality parameters have been proposed as additional information to be included in the LCI results. One classification scheme suggests water flows to be characterized as usable for drinking water purposes, agricultural activities, and industrial applications (Jolliet et al., 2003b). Another approach recommends to also record output flows which are differentiated by changes in chemical composition (Rebitzer et al., 2007). A severe drawback of existing inventory data sets is the lack of regionalized information on water flows to properly account for potential environmental impacts due to water scarcity in (semi-) arid regions.

In spite of the eminent importance of freshwater shortage and regional variation in water availability, only few methods actually provide Impact Assessment concepts. Impacts to freshwater and marine water supplies can typically be expressed in terms of quantity and quality impacts. The extraction and usage of water resources may thus be described as the reduction in the availability of water of a particular predefined quality reflecting the consequences on both quantity and quality. Owens (2002) takes up this bivalent approach and characterizes water use by appointing indicators for in-stream use (e.g., hydropower) and off-stream use (discharge of water of diminished quality), consumption (water losses, e.g., due to evaporation) and depletion (e.g., extraction from fossil groundwater reservoirs). The downstream availability of the respective water resource forms the basis for this qualitative assessment. As a quantitative method, the approaches based on the cumulative exergy demand (CExD) (Bösch et al., 2007) and the cumulative exergy extraction from the natural environment (CEENE) can also include water (Dewulf et al., 2007). The distance-to-target method of Ecological Scarcity presents a spatially differentiated way to assess freshwater use, assigning a higher weight to water-scarce countries (Frischknecht et al., 2008). These methods focus on evaluating the impacts on the freshwater resource itself. Pfister et al., 2009, in contrast, suggest a damage-oriented method for determining impacts on endpoint level and provide regionalized characterisation factors for water use. However, most LCIA methods do not consider water usage and thereby neglect the impacts from loss in availability and quality of water. Thus, further development and discussions are needed which introduce different water use types and model the relevant impact pathways from the extraction of water from natural and man-made water reservoirs with regard to the final areas of protection. A framework for assessing freshwater use following the midpoint-endpoint modelling approach has recently been elaborated (Bayart et al., submitted for publication).

8.7. Toxicity

Human and ecotoxicological impacts have been considered by some as troublesome impact categories for several political as well as scientific reasons. One has been the lack of inventory data for emissions creating data gaps (Finnveden, 2000), others are linked to the models used and related data. These issues are sometimes solved politically by deciding what should be considered as a priority based on what is generally monitored and considered to be of high concern, such as Persistent Organic Pollutants and metals. For models and data, as in other applications, this is similarly decided through recommendations and consensus on best practice. This is, however, analogous to the situation for many other impact categories.

The limited coverage of inventory data is largely a societal issue, since the knowledge on the use and fate of the thousands of chemicals is limited. The lack of knowledge gets worse when the chemical substances that are produced as unwanted side products and the entire bulk of organic emissions in the case of waterborne releases are considered (Köhler, 2006; Köhler et al., 2006), which is generally not the case in LCA studies. Case studies where extra efforts have been made to include a larger number of emissions with potential toxicological relevance have also shown that this can change the results and conclusions in current practice (Köhler, 2006; Larsen et al., 2009).

Another related problem is sometimes the lack of robust toxicological and physicochemical data necessary for the Impact Assessment. Typically characterisation factors are published for less than 2000 substances, and the current situation for the LCA practitioner who wishes to include the chemical-related impacts in the Impact Assessment is thus that: (a) there will often be many substances in the LCI for which no characterisation factor is available from any of the databases without further modelling, (b) for some substances several of the models may have published characterisation factors, but these often vary substantially between sources. While there is often a lot of effort spent on the inventory, there is traditionally little effort made by practitioners to fill these gaps or select the most appropriate factors.

While coverage is generally good for the emissions of highest political concern, and data gaps can often be filled, the coverage may not be for the most important chemical emissions in the life cycle of a specific product from an impact perspective if trying to consider everything. Again this is largely a societal problem and not specific for LCA. LCAs therefore cannot be expected to do more in these respects than what is feasible in other disciplines for data, such as chemical risk assessment. This is partly due to the lack of guidance available to fill gaps, which can often be done through very straightforward approaches based, e.g., on chemical similarity.

A third problem has been the perception of a lack of consensus concerning the characterisation methods used. A number of different models have been developed for this purpose around the world over the last 15 years (e.g., Braunschweig and Müller-Wenk, 1993; Guinée and Heijungs, 1993; Walz et al., 1996; Hauschild and Wenzel, 1998; Krewitt et al., 1998; Steen, 1999; Goedkoop and Spriensma, 2000; Huijbregts et al., 2000b; McKone and Hertwich, 2001; Crettaz et al., 2002; Pennington et al., 2002; Molander et al., 2004). A good overview of many of the models is given by Udo de Haes et al. (2002). It is important to note, however, that similar to chemical risk assessment adopted in some countries/regions, there has been a general convergence on the use of straightforward multimedia models.

Despite the convergence, available models sometimes vary in their scope, applied modelling principles, and not least in terms of the characterisation factors they produce, as revealed by comparative studies (e.g., Dreyer et al., 2003; Pant et al., 2004). The differences in these studies can obviously be also due to the way the models are parameterised and the data adopted, rather than the models themselves. This has been demonstrated with the commonly used multimedia approaches widely in the literature and over two decades.

The chemical-related impacts are, for these as well as other political reasons, often excluded from the LCIA which de facto reduces it to an energy Impact Assessment (Hauschild, 2005), which raises the question of whether any of the other parts of the LCA should be done for the same arguments and undermines the usefulness of doing an LCA. This unsatisfactory situation was the background of various activities including those of SETAC (Udo de Haes et al., 2002), the OMNITOX project (Molander et al., 2004), and later a Task Force on Toxic Impacts under the UNEP/SETAC Life Cycle Initiative.

The UNEP/SETAC group of experts analysed the prominent existing characterisation models for toxic impacts, and in collaboration between the teams behind the models created a new scientific consensus multimedia model, USEtox (http://www.usetox.org). USEtox is intended to form the basis of future recommendations from the UNEP/SETAC Life Cycle Initiative on characterisation of toxic impacts (Rosenbaum et al., 2007; Hauschild et al., 2008a). It was developed as a parsimonious multimedia chemical fate, exposure, and effect model – as simple as possible but as complex as needed meaning that in a comparison of the characterisation factors for existing characterisation models, the key features and most important choices have been identified and built into this new model (Hauschild et al., 2008a).

8.8. Indoor air

Modelling of human toxicity as described above considers exposure to outdoor air. However, indoor concentrations of chemicals and resulting human exposures often substantially exceed corresponding outdoor concentrations, mainly because of a much lower dilution volume. Moreover, people spend most of their time indoors, which in industrial countries can amount to more than 20 h a day on average. Both aspects often lead to intake fractions from indoor air of up to several orders of magnitude larger than for outdoor emissions. Therefore, neglecting indoor chemical exposure and related human health impact might lead to a major gap in a "from cradle to grave" assessment.

In spite of the relevance of indoor air exposure, only some LCIA methods consider indoor health effects. One example is the Danish Environmental Design of Industrial Products (EDIP) Method (Hauschild and Wenzel, 1998), which defines an 'exposure threshold' as 1/10 or 1/100 (depending on the effect) of the legal limit. More recent methods (Hofstetter and Norris, 2003; Schmidt et al., 2004) use statistics of occupational accidents and illnesses to estimate impacts to the working environment per industry sector. This "top-down" approach is easy to apply but is rather uncertain, because a large number of occupational diseases remain unreported. Another factor contributing to the uncertainty is the long latency periods from exposure to disease for many chemicals. Recently, first attempts have been made to integrate indoor exposure models with environmental models commonly applied within LCA. Meijer et al. (2005a,b) developed a model with which household exposure to chemicals and radiation emitted to indoor air can be assessed. Furthermore, Hellweg et al. (2005) used bulk-mixing models for occupational exposure in conjunction with multimedia models for the assessment of chemical exposure in the environment. Both studies illustrated that, indoor airflow and exposure models are, in principle, compatible with environmental models, and that the importance of health effects from indoor exposure in comparison to the total human-toxicity potential may be significant. Thus, the consideration of indoor exposure to chemicals, radiation, and particles within LCA is possible. This is also desirable, assuming that a "comprehensive" assessment tool such as LCA should not neglect potentially relevant effects to human health.

Within the first phase of the UNEP/SETAC Life Cycle Initiative, exposure models from occupational and household hygiene studies and practices were reviewed, and recommendations were provided on the appropriateness of various model alternatives in the context of LCA (Hellweg et al., 2009). For screening purposes, a one-box model, used for quantifying intake fractions, was recommended as a default model (Fig. 2). This model does not consider inhomogeneous mixing throughout the room, but it is consistent with environmental fate and exposure models, in terms of general modelling approach as well as level of model detail.

It is one of the primary goals to move this box-model approach forward and to couple indoor and environmental models within the 2nd phase of the Life Cycle Initiative (USETox model) in order to

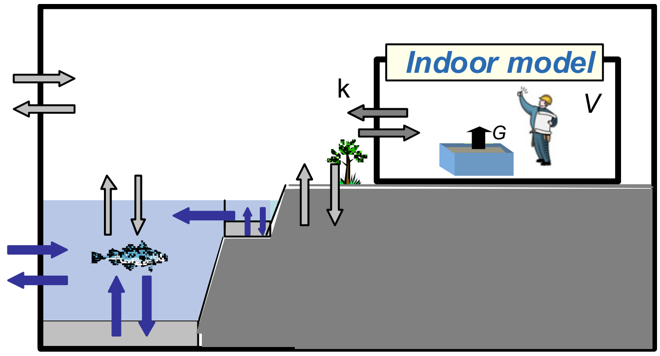

Fig. 2. Nesting a One-Box Indoor Model into common environmental fate and exposure models (G: emission rate in kg/h, V: room volume in m^3, and k: air exchange rate in 1/h).

assess indoor and outdoor exposure for a large range of chemicals using the same methodological basis. In addition to the model set up, a list of emission factors and model parameters for various household and occupational settings is currently set up. Indoor-model parameters may vary geographically, e.g., due to different ventilation practices. Also, the emission factors may be region-specific. For instance, indoor smoke from wood burning is a severe health problem of some emerging and developing countries, while it is less relevant in industrial countries. Thus, it is essential to quantify default parameter values and emissions factors for various regions in order check the sensitivity of the results and to take account of regional differences.

An important requirement for the final model is that users can adapt the model to their specific circumstances. This is of particular importance for indoor assessments, as the resulting assessment may depend largely on the specific features of the industrial plant or household. Thus, the results assessed with the characterisation factors that will be developed only give an indication of whether indoor exposure is important. If so, it may be advisable to use a more sophisticated model or to refine the model parameter values.

8.9. Normalisation

For normalisation, developments have been made with respect to more accurate, up-to-date, and complete data collection for different regions (Stranddorf et al., 2005; Bare et al., 2006; Lundie et al., 2007b; Wegener Sleeswijk et al., 2008) as well as to some method-related choices. For instance, should in a national normalisation database the emissions of the country be included as such, or should they be corrected for import and export, or even for time-lags between production and emission, as in electric equipment that will be discarded only 20 years afterwards (Wegener Sleeswijk et al., 2008)? Another example of a method-related development is the recognition that data gaps can introduce a more complicated bias for a normalized score than for an unnormalized score (Heijungs et al., 2007).

8.10. Weighting

The weighting element in LCA has always been a controversial issue, partly because this element requires the incorporation of social, political, and ethical values (cf. Finnveden, 1997). Not only are there values involved when choosing weighting factors, but also when choosing which type of weighting method to use, and also in the choice whether to use a weighting method at all (Finnveden, 1997). Despite the controversies, weighting is widely used in practice (e.g., Hansen, 1999). It is therefore important to critically review methods and data used. Evaluating weighting methods is, however, difficult since the values involved are difficult to identify and evaluate. However, all weighting methods use data and methods taken from different scientific disciplines that can and should be evaluated, and the value choices can be identified and clarified.

Methods for weighting can be classified in different ways (Finnveden et al., 2002):

- Often a distinction is made between panel methods and monetisation methods. In panel methods, a group of people are asked about their values. The common aspect of monetisation methods is that the values are expressed in a monetary measure. Sometimes also a third group is mentioned; distance-to-target methods, where the weighting factors are calculated as a function of some type of target value. However, it may be questioned if distance-to-target methods are weighting methods since the different targets are not weighted against each other (Finnveden et al., 2002).
- A distinction can be made between methods based on stated and revealed preferences. In methods based on stated preferences, people are asked about their preferences and this information is used. All panel methods and some monetisation methods are based on stated preferences. In methods based on revealed preferences, preferences revealed in different situations are used to calculate weighting factors. Some monetisation methods (such as so-called hedonic pricing) are based on revealed preferences.

Finally a distinction can be made between mid-point methods and end-point or damage methods as discussed above.

Several weighting methods that were developed during the 1990s are still used. Examples include the EPS method (Steen, 1999), which is an end-point method based on monetary measures. Another example is the Ecoindicator'99 (Goedkoop and Spriensma, 2000) which also is an end-point method, however, based on a panel approach. Other examples include distance-to-target methods EDIP (Wenzel et al., 1997) and Ecoscarcity (Frischknecht et al., 2008).

Recently several methods have also been published in the scientific literature increasing the credibility and the review process. Examples of more recent methods are the LIME weighting method based on monetary valuation of end-points (Itsubo et al., 2004; Weidema, 2009), the Ecotax method based on a monetary valuation of mid-points (Finnveden et al., 2006), the BEPAS method also based on a monetary valuation of mid-points (Zhang et al., 2006), and panel methods for mid-points (Soares et al., 2006; Huppes et al., 2007). Methodological aspects of panel methods are also discussed by Mettier and Hofstetter (2005).

Under the heading weighting methods, also different types of proxy methods can be considered which focus on one or a few aspects which are then considered as being indicative of the whole result. Examples of such methods include the cumulative fossil energy demand (Huijbregts et al., 2006) and the ecological footprint (Huijbregts et al., 2008).

9. Uncertainties in LCA

As with many decision support tools, uncertainties are often not considered in LCA studies although they can be high. But, there is arguably a necessity for an analysis of the uncertainties involved in carrying out an LCA study to help focus research efforts and also to provide support in the interpretation of LCA study results. In this section, we build on and extend several review papers that concentrate on uncertainty in LCA, notably Huijbregts et al. (2001), Björklund (2002), Heijungs and Huijbregts (2004), Geisler et al. (2005), and Lloyd and Ries (2007).

Uncertainty can be defined in many ways, but one definition that appears to be useful in the present context is: "the discrepancy between a measured or calculated quantity and the true value of that quantity". There are various classifications of uncertainty in the literature (Funtowicz and Ravetz, 1990; Morgan and Henrion, 1990; Huijbregts, 1998). Examples are: data uncertainty, model uncertainty, variability, epistemic uncertainty, etc. Here, we will distinguish between sources and types of uncertainties. Sources are the input elements of LCA that may be uncertain, and types reflect the different aspects that may be "wrong" with these sources. In LCA, we may distinguish the following sources:

- data, e.g., electricity use of a heating boiler, CO_2 emissions from a coal fired power plant, and GWP of dinitrogenoxide;
- choices, e.g., system boundaries, allocation principles, and time horizon in Impact Assessment;
- relations, e.g., the linear dependence of travelled distance on fuel input, the linear dependence of acidification on SO_2 emissions, and the discounting formula used for long-term impacts.

The types of uncertainties are partly related to these sources, as illustrated by these examples:

- data can show variability, e.g., the electricity use of various similar boilers may be different, and even for one and the same boiler it may change over time or depend on the conditions;
- data can be miss-specified, e.g., instead of the electricity use of a 80-l boiler in France in 2007, one may have data for a 75-l boiler in Germany in 2006;
- data may be erroneous, e.g., a typo may have been made, a mistake in the units may have occurred, or a decimal point may have been confused for a thousands separator;
- data may be incomplete, e.g., one may lack the emissions of dichlorobenzene from a certain incinerator;
- data may be subject to round-off, e.g., 0.342 may have been entered as 0.3, which is more than 10% wrong;
- choices may have been made inconsistently with the goal and scope of the analysis, e.g., average technology for certain processes instead of best available technology;
- choices may have been made inconsistently across alternatives, e.g., different allocation methods for different product chains;
- relations may be wrong, e.g., a linear dependence of acidification on SO_2 emissions may not reflect the true relationship;
- relations may be incomplete, e.g., the influence of background levels of contaminants may have been neglected;
- relations may have been implemented inaccurately in software, e.g., matrix inversion routines may be sensitive for the choice of algorithm.

It will be clear that there are many types of uncertainties, and that many of these types will show up in a typical LCA.

When LCA is used for decision support, as with any tool, the uncertainty of the results can be an important part of the information. In the ISO 14040 framework, issues of uncertainty are mentioned as part of the LCI and LCIA phases, but most prominently as a part of the Interpretation phase. Although the standard mentions some issues related to uncertainty analysis, no concrete guidance is provided. This is, however, being addressed by several of the ongoing initiatives that complement ISO and provide more explicit recommendations.

Uncertainty can be dealt with in several ways, here we distinguish between the "scientific" way, the "social way", and the "statistical way". The "scientific" way to deal with large uncertainties (that sometimes prevent results being used in decision support) is to do more research: find better data, make better models, etc. Although we do not deny the usefulness of finding better data and making better models, the value is restricted in many cases where decisions will be made anyway. The urgency of environmental questions often requires swift action and forbids decision makers to wait for complete evidence on all matters. The disappearance of a species is only proven when it is too late.

The "social" way is to discuss the uncertain issues with stakeholders and to find consensus on data and choices. A variant of this is the "legal" way, in which an authoritative body establishes the data, models, and choices. For instance, the European Commission in combination with other governments, industry, UNEP, as well as various scientific advisory bodies is currently in the process of establishing formal recommendations for LCA. Although consensus, recommendations, and policy are admittedly important ingredients of modern society, they have on occasions been in sharp conflict with science and rational inquiry, suggesting a need for caution. One remedy may sometimes be an even broader input from stakeholders, but this has the danger of enlarging the confusion between facts and opinions even more (Heijungs, 1999). On the other hand, harmonization and recommendations can also provide a reference for further scientific discussions and development.

The "statistical" way, in contrast to the previous two ways, does not try to remove or reduce the uncertainty, but to incorporate it. Statistical theory comprises a large body of methods to do:

- *parameter variation and scenario analysis*: these involve calculating a result with a number of different data values and/or choices, e.g., using the maximum and the minimum fuel efficiency, and seeing if the results are stable;
- classical statistical theory on the basis of probability distributions, tests of hypothesis, etc.;
- Monte Carlo simulations, bootstrapping, and other sampling approaches;
- using analytical methods, based on first-order error propagation;
- using less conventional methods, such as non-parametric statistics, Bayesian analysis, and fuzzy set theory;
- using qualitative uncertainty methods, for instance, based on data quality indicators.

In the present-day LCA studies uncertainties are increasingly taken into account, although the coverage remains limited typically to parameter uncertainty. Some databases (e.g., the ecoinvent inventory data) contain probability distributions for almost all data items. Most of the major tools for LCA contain algorithms for conducting Monte Carlo analysis, while some programs enable the use of fuzzy methods or of analytical approaches.

Especially on fuzzy set theory, advances have been reported the last few years by, amongst others Mauris et al. (2001), Benetto et al. (2006), Seppälä (2007), and Tan (2008). The advance on Bayesian methods in LCA has been much smaller, although some progress has been made (e.g., by Lo et al., 2005).

Another relevant development is that a link has been established between interpretation and goal and scope definition, in a broadening of the use of scenarios from defining the functional unit or specifying the technological conditions to varying these scenarios in an uncertainty perspective (Spielmann et al., 2005).

To facilitate systematic uncertainty analysis, there is a need for standardization of uncertainty information (Heijungs and Frischknecht, 2005). The existence of the ISO standard for expressing uncertainties (the guide to the expression of uncertainty in measurement, or GUM for short (Anon., 1993)) appears to have escaped notice of those that were involved of designing, e.g., the ISO standards for LCA. The reporting of uncertainties in data sources and LCA results is likely to improve significantly if the LCA community adheres to the principles outlined in GUM.

We have indicated that the unique feature of the "statistical" approach is to deal with uncertainty that it does not remove or reduce uncertainty, but rather incorporates it. Nevertheless, statistical approaches may have a second useful function in reducing uncertainties. One example is provided by the data reconciliation approaches on the basis of fuzzy uncertainties (Tan et al., 2007). Another example is the use of key issue analysis (Heijungs, 1996) (also known as sensitivity analysis) to priorities places for carrying on the "scientific approach", i.e., collecting better data.

The above uncertainty was defined in terms of the discrepancy between a measured or calculated quantity and the true value of that quantity. This brings us to the question of validation, the comparison of measurement or calculation and "truth". Ciroth and Becker (2006, p. 297) argue that "validation in LCA models offers tremendous possibilities for model improvements as well as improvements of the quality of decisions supported by LCA models". On the other hand, it has been argued that validation of models like LCA is impossible, on different grounds. To Oreskes et al. (1994) and Heijungs (2001), the only possible validation is the piecemeal one: unit processes, steps in impact pathways, etc., each building block may be validated separately, and as long as the gluing together proceeds according to strict procedures and mathematical rules, we can hope that LCA makes sense after all. Clearly, the area of validation as well as the larger area of uncertainty in LCA need further attention and development.

10. Discussion

The LCA methodological development has been strong over the last decades. Several of the limitations that have been mentioned in critical reviews have also been addressed. Several of them can never be fully solved and are common to other tools, but better data and better methods are developed. For example:

- LCA is very data intensive, and lack of data can restrict the conclusions that can be drawn from a specific study. However, as discussed above, better databases are developed, and growing experience will tell us where to focus our efforts including in terms of quality-assurance. The development of hybrid IO-LCA models is interesting in this aspect, since IOA can provide us with data for the whole economic system but this also involves some fundamental approximations that need to be better discussed. If judged suitable, it is then always possible to find some data as long as price information is available. This may mean that we can move from a discussion on data gaps to a discussion about data quality and uncertainty, at least for inventory. For specific substances, the risk of data gaps will, however, always remain as long as societies are handling thousands of chemical products with limited knowledge on their use and fate combined with the lack of guidance for dealing with data gaps from the scientific community. Additionally, LCA practitioners may have to dedicate the same resources to LCIA as other parts of a study including to fill gaps.
- LCA aims at providing a comprehensive view of environmental impacts. However, not all types of impacts are equally well covered in a typical LCA. For example, the methods for the Impact Assessment of land use, including impacts on biodiversity, and resource aspects, including freshwater resources, are problematic and need to be improved. Improvements can be made through further interaction with related fields. Several methods have been described in the literature and these need

to be tested and evaluated. Also human health aspects other than those related to outdoor exposure of pollutants have traditionally received limited attention. The development of indoor exposure models and the possible inclusion of other health aspects in social LCA will improve the situation. Last, water use may have tremendous environmental impact in water-scarce countries and should not remain omitted in LCA. These are examples of developments that can make future assessments more comprehensive and more reliable.

- An LCA can include several methodological choices which are uncertain and may potentially influence the results. Examples include allocation methods and time limits for the Inventory Analysis, and choices of characterisation methods for the Impact Assessment. Questions related to system boundaries and methodological choices are common for all systems analysis tools, and the lack of a "right" answer can sometimes be problematic. However, two trends are very instrumental in this respect. The first is the increased understanding of the connections between the aims of the study, the questions being asked and the choices made. For example, if the aim of a study is to assess the consequences of a choice, the data used and the system boundaries chosen should reflect these consequences. The data and system boundaries used can then be discussed and assessed in relation to their appropriateness for this specific aim. In such situations, some data and system boundaries will be more appropriate and therefore the uncertainty due to methodological choices is reduced. Another trend is the harmonization and consensus building that has occurred and is still ongoing. This is largely based on a scientific discussion on what from a scientific point of view is regarded as the most appropriate choices. For the Impact Assessment this includes the work on developing best practice. The developments for human and ecotoxicity is here of special interest. The recent developments for abiotic resources may also lead to an increased harmonization.
- For all quantitative methods, including LCA, different types of uncertainties are an important issue. The development of tools to properly handle the uncertainty was described above. A special type of uncertainty is related to lack of knowledge on the actual system to be studied. This is the case, for example, for future systems, since the future is inherently uncertain. The systematic use of different types of scenarios can be instrumental in handling this type of uncertainty. The development of generic future scenarios that can be used in different studies could help in studying the possible influence on the results in particular studies from different possible developments.
- The world is multi-dimensional. This means that sometimes different aspects have to be weighted and valued against each other. There has sometimes been reluctance to discussing weighting methods within the LCA standardization and harmonization working groups. This is understandable from the point of view that the values cannot be harmonized and there is no way we can find out which values are "right", from a scientific point of view. However, methods and data used in the weighting methods can and should be discussed and evaluated for consistency by scientific methods. The development of weighting methods to be used in LCA has benefited from developments within environmental economics and multi-criteria decision analysis. It is also important that weighting methods for LCA continue to be published in the scientific literature after normal review processing for increased credibility and check of data and methods.

The growing confidence in LCA and life cycle thinking is illustrated in its increased use in different parts of the society, and enhanced through the development of recommendations by authoritative bodies. Industry has used LCA for a long time both for decision-making and learning (e.g., Berkhout and Howes, 1997; Hansen, 1999; Frankl and Rubik, 2000). However, there are some new trends in applications emerging. One is the interest in life cycle thinking from retailers. This is illustrated by, e.g., Wal-Mart who set up targets for reduced environmental impacts of products they are selling and require life-cycle related information from their suppliers (Wal-Mart, 2007, 2008). Additionally, sales of LCA software tools increased tremendously due to a rising demand in life-cycle-related environmental information over the last years (Pré Consultants, 2008).

Another trend is the increased use of LCA and promotion of life cycle thinking on a policy level. For example, life cycle thinking is an important element of European environmental policy. In June 2003, the European Commission's Integrated Product Policy (IPP) Communication (European Commission, 2003) aimed at improving the environmental performance of products (both goods and services) throughout their life cycles ("cradle to grave"), committing to several activities to promote LCA and to help address political as well as scientific barriers that exist. In December 2005, the important role of life cycle thinking was further strengthened in the Commission's Thematic Strategies on the Sustainable Use of Natural Resources (European Commission, 2005a) and on the Prevention and Recycling of Waste (European Commission, 2005b). In 2008, these activities are being further strengthened through the Sustainable Consumption and Production Action Plan (European Commission, 2008c). Another example is the proposal for a directive on the use of energy from renewable sources (European Commission, 2008a) where savings of greenhouse gas emissions from the use of renewable fuels are to be calculated with a clear life-cycle perspective. On the national level, such legislation is already in place in some countries as well as other examples addressing emerging issues of high political concern. For instance, a new law in Switzerland requires a complete LCA of biofuels in order to quantify the fuel tax to be paid.

From a methodological perspective, the combination of IOA and LCA is promising to complement LCA applications for macro-level policy support. Recent research studies in connection with IPP helped broaden the areas of LCA type applications, identifying which sectors may have the highest environmental impacts.

Hybrid IOA-LCA is one example of how LCA can be combined with other environmental systems analysis tools. Other examples are the use of monetary weighting methods in combination with Cost-Benefit Analysis and Life Cycle Costing (e.g., Carlsson Reich, 2005) and the combination of LCA and Strategic Environmental Assessment (SEA). The latter is a procedural tool for assessing environmental impacts of plans, policies, and programs. LCA may be used as an analytical tool within the framework of SEA (Nilsson et al., 2005; Salhofer et al., 2006).

While there have been many achievements, including, for example, the international standards on LCA (ISO, 2006a,b), and LCA is recognized as the best tool for assessing the life cycle impacts of products, there are still barriers that inhibit the broader implementation of life cycle thinking (European Commission, 2003). Some of these barriers may be reduced with further research. Referring to the discussions on attributional and consequential, the handling of time, space, and economic and social issues in LCA, it has become clear that there is a need for structuring the varying fields of LCA and drafting specific research programs for this. The EU 6th Framework Co-ordination Action CALCAS (2009) aims to achieve this.

A wealth of methods and data is available, and government bodies as well as international business representatives feel that there is a need for guidance on what to use and when. For this purpose, several international activities have been initiated including the UNEP/SETAC Life Cycle Initiative (UNEP, 2002) and the International Reference Life Cycle Data System (ILCD) (European Commission, 2008b). The "Life Cycle Initiative" has also been instrumental in supporting LCA activities in emerging economies. This is important since increasing environmental impacts occur in quickly growing economies. Life-cycle thinking in all countries is important to avoid shifting of burdens between countries and impacts. Urgent action in many countries is critical for sustainable development. Supporting the development and dissemination of practical and quality-assured LCA tools worldwide is thus an important path to follow.

11. Conclusions

Environmental considerations need to be integrated in many types of decisions. This includes decisions related to goods and services. In order to do that, knowledge must be available. When studying environmental impacts of products and services it is vital to study these in a life cycle perspective, in order to avoid problem-shifting from one part of the life-cycle to another, from one geographical area to another. It is also important to make a comprehensive assessment in terms of environmental problems in order to avoid problem-shifting from one area of environmental concern to another. Life Cycle Assessment aims at making a comprehensive assessment of the environmental impacts of products and services in a life-cycle perspective.

The LCA methodology has developed and somewhat matured during the last decades. Current activities regarding databases, quality assurance, consistency, and harmonization of methods contribute to this. It is also interesting to note the development of new application areas indicating the need to assess and communicate environmental impacts of products.

The review presented in this paper indicates several areas where the development has been strong during the last years. These include a better understanding of the difference between attributional and consequential LCA, methods for hybrid IOA-LCA, better models for impact assessment, and databases for the inventory analysis. There are at the same time several areas where further development would be useful. Examples of such areas include further development of tools for consequential LCA, of methods for assessment of impacts on ecosystem services from land use and impacts from water use, and weighting methods. Further development and maintenance of databases should also be a prioritised area.

Disclaimer

The views presented in this manuscript are those of the authors and do not necessarily reflect the opinions of the associated organisations.

Acknowledgements

The work presented here is financially supported by several organisations including the European Commission (through the CALCAS project), the Foundation for Strategic Environmental Research (MISTRA), and the Swedish Environmental Protection Agency (through the research programme Towards Sustainable Waste Management). All are gratefully acknowledged.

References

Anon, 1993. Guide to the Expression of Uncertainty in Measurement. ISO, Geneva.
APME, 2007. Plastics Europe. Association of Plastics Manufacturers (APME). Life Cycle and Eco-profiles. http://www.plasticseurope.org/Content/Default.asp?PageID =392#.
Åström, S., 2004. Marginal Production of Pulpwood to the Pulp and Paper Industry. Master's Thesis T2004-283, Department of Energy Technology, Chalmers University of Technology, Gothenburg, Sweden.
Audsley, E., Alber, S., Clift, R., Cowell, S., Crettaz, P., Gaillard, G., Hausheer, J., Jolliet, O., Kleijn, R., Mortensen, B., Pearce, D., Roger, E., Teulon, H., Weidema, B., van Zeijts, H., 1994. Harmonisation of environmental life cycle assessment for agriculture. Final Report of EU Concerted Action AIR3-CT94-2028, Brussels.
Ayres, R.U., 1995. Life cycle analysis: a critique. Resour. Conserv. Recycling 14, 199–223.
Bare, J.C., Hofstetter, P., Pennington, D., Udo de Haes, H.A., 2000. Midpoints versus endpoints: the sacrifices and benefits. Int. J. LCA 5 (6), 319–326.
Bare, J.C., Norris, G.A., Pennington, D.W., McKone, T.E., 2003. TRACI, the tool for the reduction and assessment of chemical and other environmental impacts. J. Ind. Ecol 6 (3–4), 49–78.
Bare, J.C., Gloria, T.P., Norris, G., 2006. Development of the method and US normalization database for life cycle impact assessment and sustainability metrics. Environ. Sci. Technol 40, 5108–5115.
Baumann, H., Tillman, A.-M., 2004. A Hitchhiker's Guide to Life Cycle Assessment. Studentlitteratur, Lund.
Bayart, J.B., Bulle, C., Deschênes, L., Margni, M., Pfister, S., Vince, F., Koehler, A., submitted for publication. A framework for assessing off-stream freshwater use in LCA. Int. J. LCA.
Benetto, E., Dujet, C., Rousseaux, P., 2006. Possibility theory: a new approach to uncertainty analysis? Int. J. LCA 11 (2), 114–116.
Berkhout, F., Howes, R., 1997. The adoption of life-cycle approaches by industry: patterns and impacts. Resour. Conserv. Recycling 20, 71–94.
Birkved, M., Hauschild, M., 2006. PestLCI – a model for estimating field emissions of pesticides in agricultural LCA. Ecol. Model 198 (3–4), 433–451.
Björklund, A., Finnveden, G., 2005. Recycling revisited – life cycle comparisons of waste management strategies. Resour. Conserv. Recycling 44, 309–317.
Björklund, A., Carlsson, A., Finnveden, G., Palm, V., Wadeskog, A., 2007. IPP-indicators for private and public consumption basedon environmental accounts and LCA. TRITA-INFRA-FMS 2007:7.KTH, Stockholm.
Björklund, A.E., 2002. Survey of approaches to improve reliability in LCA. Int. J. LCA 7 (2), 64–72.
Börjeson, L., Höjer, M., Dreborg, K.-H., Ekvall, T., Finnveden, G., 2006. Scenario types and techniques – towards a user's guide. Futures 34, 723–739.
Bösch, M.E., Hellweg, S., Huijbregts, M., Frischknecht, R., 2007. Applying cumulative exergy demand (CExD) indicators to the ecoinvent database. Int. J. LCA 12, 181–190.
Braunschweig, A., Müller-Wenk, R., 1993. Ökobilanzen für Unternehmungen; eine Wegleitung für die Praxis. Verlag Paul Haupt, Bern.
Bruch, K.H., Gohlke, D., Kögler, C., Krüger, J., Reuter, M., Röpenack, I.v., Rombach, E., Rombach, G., Winkler, P., 1995. Sachbilanz einer Ökobilanz der Kupfererzeugung und -verarbeitung, Part 1. Metall 49 (4), 252–257.
CALCAS, 2009. Co-ordination action for innovation in life-cycle analysis for sustainability. <http://fr1.estis.net/sites/calcas/default.asp>.
Carlsson Reich, M., 2005. Economic assessment of municipal waste management systems – case studies using a combination of life cycle assessment (LCA) and life cycle costing (LCC). J. Cleaner Prod 13, 253–263.
Capello, C., Hellweg, S., Badertscher, B., Hungerbühler, K., 2005. Life-cycle inventory of waste solvent distillation: statistical analysis of empirical data. Environ. Sci. Technol. 39 (15), 5885–5892.
Christensen, T.H., Bhander, G., Lindvall, H., Larsen, A.W., Fruergaard, T., Damgaard, A., Manfredi, S., Baldrin, A., Riber, C., Hauschild, M., 2007. Experience with the use of LCA-modelling (EASEWASTE) in waste management. Waste Manage. Res. 25, 257–262.
Ciroth, A., 2007. ICT for environment in Life Cycle Applications: open LCA – a new open source software for Life Cycle Assessment. Int. J. LCA 12 (4), 209–210.
Ciroth, A., Becker, H., 2006. Validation. The missing link in life cycle assessment. Towards pragmatic LCAs. Int. J. LCA 11, 295–297.
Claeson, U., 2000. Analyzing Technological Change Using Experience Curves – A Study of the Combined Cycle Gas Turbine Technology. LicEng thesis, Chalmers University of Technology, Gothenburg, Sweden.
Consoli, F., Allen, D., Boustead, I., Fava, J., Franklin, W., Jensen, A.A., de Oude, N., Parrish, R., Perriman, R., Postlethwaite, D., Quay, B., Siéguin, J., Vigon, B. (Eds.), 1993. Guidelines for Life Cycle Assessment. A Code of Practice. SETAC Press, Pensacola, FL.
CPM, 2007. SPINE@CPM database. Competence Center in Environmental Assessment of Product and Material Systems (CPM), Chalmers University of Technology, Göteborg.
Crettaz, P., Pennington, D., Rhomberg, L., Brand, K., Jolliet, O., 2002. Assessing human health response in life cycle assessment using ED10s and DALYs: Part 1 – Cancer effects. Risk. Anal. 22 (5), 939–946.
Curran, M.A., 2007. Co-product and input allocation approaches for creating life-cycle inventory data: a literature review. Int. J. LCA 1/07 13, 65–78 (special issue).
Curran, M.A., Mann, M., Norris, G., 2005. The international workshop on electricity data for life cycle inventories. J. Cleaner Prod. 13 (8), 853–862.
den Boer, J., den Boer, E., Jager, J., 2007. LCA-IWM: a decision support tool for sustainability assessment of waste management systems. Waste. Manage. 27, 1032–1045.
Deutsches Kupferinstitut, 1995. Sachbilanz einer Ökobilanz der Kupfererzeugung und – verarbeitung. Deutsches Kupferinstitut, Düsseldorf.
Dewulf, J., Bösch, M.E., De Meester, B., Van der Vorst, G., Van Langenhove, H., Hellweg, S., Huijbregts, M.A.J., 2007. Cumulative Exergy Extraction from the

Natural Environment (CEENE): a comprehensive Life Cycle Impact Assessment method for resource accounting. Environ. Sci. Technol. 41, 8477–8483.
Doka, G., 2003. Life Cycle Inventories of Waste Treatment Services. Ecoinvent Report No. 13 (Parts I–IV), Swiss Centre for Life Cycle Inventories, Dübendorf.
Doka, G., Hischier, R., 2005. Waste treatment and assessment of long-term emissions. Int. J. LCA 10, 77–84.
Dreyer, L.C., Niemann, A.L., Hauschild, M.Z., 2003. Comparison of three different LCIA methods: EDIP97, CML2001 and Eco-indicator 99. Does it matter which one you choose? Int. J. LCA 8 (4), 191–200.
EAA, 2007. European Aluminium Association (EAA). Environmental Profile Report and LCA Data. <http://www.eaa.net/eaa/index.jsp> (data available on request only).
Ecoinvent, 2007. Swiss Centre for Life Cycle Inventories (Ecoinvent Centre). Ecoinvent Database. Ecoinvent Centre, Dübendorf, 2004 and 2007. <http://www. ecoinvent.org>.
Ehrenfeld, J.R., 1998. The importance of LCA – Warts and all. J. Ind. Ecol 1 (2), 41–49.
Ekvall, T., 1999. System Expansion and Allocation in Life Cycle Assessment. Ph.D. Thesis, Chalmers University of Technology, Gothenburg, Sweden.
Ekvall, T., 2000. A market-based approach to allocation at open-loop recycling. Resour. Conserv. Recycling 29 (1–2), 93–111.
Ekvall, T., Andrae, A., 2006. Attributional and consequential environmental assessment of the shift to lead-free solders. Int. J. LCA 11 (5), 344–353.
Ekvall, T., Finnveden, G., 2001. Allocation in ISO 14041 – a critical review. J. Cleaner Prod 9, 197–208.
Ekvall, T., Weidema, B.P., 2004. System boundaries and input data in consequential life cycle inventory analysis. Int. J. LCA 9 (3), 161–171.
Ekvall, T., Tillman, A.-M., Molander, S., 2005. Normative ethics and methodology for life cycle assessment. J. Cleaner Prod 13 (13–14), 1225–1234.
Ekvall, T., Mattsson, N., Münter, M., 2006. Consequential modelling of Vistar combustion: a feasibility study. In: Abstracts – 16th Annual Meeting of SETAC – Europe, The Hague, The Netherlands, May 2006, p. 281.
Elkington, J., 1998. Cannibals with Forks – The Triple Bottom Line of 21st Century Business. New Society Publishers, Canada.
Eriksson, O., Frostell, B., Björklund, A., Assefa, G., Sundqvist, J.-O., Granath, J., Carlsson, M., Baky, A., Thyselius, L., 2002. ORWARE – a simulation tool for waste management. Resour. Conserv. Recycling 36, 287–307.
Eriksson, O., Finnveden, G., Ekvall, T., Björklund, A., 2007. Life Cycle Assessment of fuels for district heating: a comparison of waste incineration, biomass- and natural gas combustion. Energy Policy 35, 1346–1362.
European Commission, 2003. Integrated Product Policy Communication. COM (2003) 302 final.
European Commission, 2005a. Thematic Strategy on the Sustainable Use of Natural Resources. COM (2005) 670 final.
European Commission, 2005b. Thematic Strategy on the Prevention and Recycling of Waste. COM (2005) 666 final.
European Commission, 2007a. European Commission, Directorate General Joint Research Centre (JRC), European Reference Life Cycle Database (ELCD). <http://lca.jrc.ec.europa.eu/lcainfohub/>.
European Commission, 2008a. Proposal for a Directive of the European Parliament and of the Council on the Promotion of the Use of Energy from Renewable Sources, Version 15.4. Commission of the European Communities, Brussels.
European Commission, 2008b. European Platform on Life Cycle Assessment. <http://lca.jrc.ec.europa.eu/>.
European Commission, 2008c. Sustainable Consumption and Production Action Plan (SCP). COM (2008), 397. <http://ec.europa.eu/environment/eussd/escp_en.htm>. final.
Fava, J., Consoli, F., Denison, R., Dickson, K., Mohin, T., Vigon, B. (Eds.), 1993. A Conceptual Framework for Life Cycle Impact Assessment. SETAC Press, Pensacola, FL.
FEFCO, 2006. European Federation of Corrugated Board Manufacturers (FEFCO). European Database for Corrugated Board – Life Cycle Studies, FEFCO, Brussels, Belgium.
Finnveden, G., 1997. Valuation methods within LCA – where are the values? Int. J. LCA 2, 163–169.
Finnveden, G., 2000. On the limitations of life cycle assessment and environmental systems analysis tools in general. Int. J. LCA 5, 229–238.
Finnveden, G., 2005. The resource debate needs to continue. Int. J. LCA 10, 372.
Finnveden, G., 2008. A world with CO_2-caps. Electricity production in consequential assessments. Int. J. LCA 13, 365–367.
Finnveden, G., Moberg, Å, 2005. Environmental systems analysis tools – an overview. J. Cleaner Prod. 13, 1165–1173.
Finnveden, G., Nilsson, M., 2005. Site-dependent life cycle impact assessment in Sweden. Int. J. LCA 10, 235–239.
Finnveden, G., Östlund, P., 1997. Exergies of natural resources in life cycle assessment and other applications. Energy 22, 923–931.
Finnveden, G., Albertsson, A.C., Berendson, J., Eriksson, E., Höglund, L.O., Karlsson, S., Sundqvist, J.-O., 1995. Solid waste treatment within the framework of life-cycle assessment. J. Cleaner Prod. 3, 189–199.
Finnveden, G., Hofstetter, P., Bare, J., Basson, L., Ciroth, A., Mettier, T., Seppälä, J., Johansson, J., Norris, G., Volkwein, S., 2002. Normalization, grouping and weighting in life cycle impact assessment. In: Udo de Haes, et al. (Eds.), Life-Cycle Impact Assessment: Striving Towards Best Practise. SETAC Press, Pensacola, FL, pp. 177–208.
Finnveden, G., Eldh, P., Johansson, J., 2006. Weighting in LCA based on ecotaxes – development of a mid-point method and experiences from case studies. Int. J. LCA 11 (1), 81–88 (special issue).
Format Converter, 2008. Format conversion tool. <http://sourceforge.net/projects/openlca>.
Frankl, P., Rubik, F., 2000. Life Cycle Assessment in Industry and Business. Adoptions, Patterns, Applications and Implications, Springer Verlag, Heidelberg.
Friedman, M., 1976. Price Theory. Aldine, Chicago, USA.
Frischknecht, R., Althaus, H.-J., Bauer, C., Doka, G., Heck, T., Jungbluth, N., Kellenberger, D., Nemecek, T., 2007. The environmental relevance of capital goods in life cycle assessments of products and services. Int. J. LCA 13, 7–17 (special issue).
Frischknecht, R., Steiner, R., Jungblut N., 2008. Ökobilanzen: Methode der ökologischen Knappheit – Ökofaktoren 2006 (Swiss Ecological Scarcity Method), Öbu – Netzwerk für Nachhaltiges Wirtschaften, Öbu SR 28/2008, Zürich.
Funtowicz, S.O., Ravetz, J.R., 1990. Uncertainty and Quality in Science for Policy. Kluwer Academic Publishers, Dordrecht.
Gäbel, K., Forsberg, P., Tillman, A.-M., 2004. The design and building of a lifecycle based process model for simulating environmental performance, product performance and cost in cement manufacturing. J. Clean. Prod. 12, 77–93.
GDI (Green Design Institute), 2007. Environmental Input–Output LCA database. <http://www.eio-lca.net> (accessed in July 2007).
Geisler, G., Hofstetter, T.B., Hungerbühler, K., 2004. Production of fine and speciality chemicals: procedure for the estimation of LCIs. Int. J. LCA 9, 101–113.
Geisler, G., Hellweg, S., Hungerbühler, K., 2005. Uncertainty analysis in life cycle assessment (LCA): case study on plant-protection products and implications for decision making. Int. J. Life Cycle Assess. 10, 184–192.
Goedkoop, M., Spriensma, R., 2000. The Eco-indicator 99 – A Damage-oriented Method for Life Cycle Impact Assessment. Methodology Report, second ed., 17-4-2000. Pré Consultants, B.V. Amersfoort, The Netherlands.
Grießhammer, R., Benoît, C., Dreyer, L.C., Flysjö, A., Manhart, A., Mazijn, B., Méthot, A.-L., Weidema, B.P., 2006. Feasibility Study: Integration of Social Aspects into LCA. UNEP-SETAC Life Cycle Initiative, Paris.
Guinée, J.B., Heijungs, R., 1993. A proposal for the classification of toxic substances within the framework of Life Cycle Assessment of products. Chemosphere 26 (10), 1925–1944.
Guinée, J.B., Udo de Haes, H.A., Huppes, G., 1993a. Quantitative life cycle assessment of products: 1. Goal definition and inventory. J. Clean. Prod. 1, 3–13.
Guinée, J.B., Heijungs, R., Udo de Haes, H.A., Huppes, G., 1993b. Quantitative life cycle assessment of products: 2. Classification, valuation and improvement analysis. J. Clean. Prod. 1, 3–13.
Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H.A., de Bruijn, J.A., van Duin, R., Huijbregts, M.A.J., 2002. Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards. Series: Eco-efficiency in Industry and Science. Kluwer Academic Publishers, Dordrecht.
Guinée, J.B., Heijungs, R., Huppes, G., 2004. Economic allocation: examples and derived decision tree. Int. J. LCA 9, 23–33.
Hansen, O.J., 1999. Status of Life Cycle Assessment (LCA) activities in the Nordic region. Int. J. LCA 44, 315–320.
Hauschild, M.Z., 2005. Assessing environmental impacts in a life cycle perspective. Environ. Sci. Technol. 39, 81A–88A.
Hauschild, M.Z., Potting, J., 2005. Spatial differentiation in life cycle impact assessment – the EDIP2003 methodology. Environmental News No. 80. The Danish Ministry of the Environment, 2005. Environmental Protection Agency, Copenhagen. <http://www.mst.dk>.
Hauschild, M.Z., Pennington, D., 2002. Indicators for ecotoxicity in life cycle impact assessment. In: Udo de Haes, H.A., Finnveden, G., Goedkoop, M., Hauschild, M., Hertwich, E., Hofstetter, P., Klöpffer, W., Krewitt, W., Lindeijer, E., Jolliet, O., Müller-Wenk, R., Olsen, S.I., Pennington, D., Potting, J., Steen, B. (Eds.), Life Cycle Impact Assessment: Striving Towards Best Practice. SETAC Press, Pensacola, FL (Chapter 6).
Hauschild, M.Z., Wenzel, H., 1998. Environmental assessment of products. In: Scientific background, vol. 2. Chapman & Hall, Kluwer Academic Publishers, United Kingdom, Hingham, MA, USA, 565 pp.
Hauschild, M.Z., Huijbregts, M., Jolliet, O., MacLeod, M., Margni, M., van de Meent, D., Rosenbaum, R.K., McKone, T., 2008a. Building a consensus model for life cycle impact assessment of chemicals: the search for harmony and parsimony. Environ. Sci. Technol. 42, 7032–7037.
Hauschild, M.Z., Olsen, S.I., Hansen, E., Schmidt, A., 2008b. Gone… but not away – addressing the problem of long-term impacts from landfills in LCA. Int. J. Life Cycle Assess. 13, 547–554.
Heijungs, R., 1994. A generic method for the identification of options for cleaner products. Ecol. Economics 10, 69–81.
Heijungs, R., 1996. Identification of key issues for further investigation in improving the reliability of life-cycle assessments. J. Cleaner Prod. 4, 159–166.
Heijungs, R., 1999. Does a participatory approach to life cycle assessment solve its problems? SETAC-Europe News 10 (5), 9–10.
Heijungs, R., 2001. A Theory of the Environment and Economic Systems. A Unified Framework for Ecological Economics and Decision-Support. Edward Elgar, Cheltenham.
Heijungs, R., Frischknecht, R., 2005. Representing statistical distributions for uncertain parameters in LCA. Relationships between mathematical forms, their representation in EcoSpold, and their representation in CMLCA. Int. J. Life Cycle Assess. 10, 248–254.
Heijungs, R., Guinée, J.B., 2007. Allocation and "what-if" scenarios in life cycle assessment of waste management systems. Waste Manage. 27, 997–1005.

Heijungs, R., Huijbregts, M.A.J., 2004. A review of approaches to treat uncertainties in LCA. In: Pahl-Wostl, C., Schmidt, S., Rizzoli, A.E., Jakeman, A.J. (Eds), Complexity and Integrated Resources Management, Transactions of the 2nd Biennial Meeting of the International Environmental Modelling and Software Society, June 2004, vol. 1. iEMSs, pp. 332–339.
Heijungs, R., Suh, S., 2002. The Computational Structure of Life Cycle Assessment. Kluwer, Dordrecht.
Heijungs, R., Guinée, J., Huppes, G., Lankreijer, R.M., Udo de Haes, H.A., Wegener Sleeswijk, A., Ansems, A.M.M., Eggels, P.G., van Duin, R., de Goede, H.P., 1992. Environmental Life Cycle Assessment of Products. Guide. Report No. 9266, CML, Leiden University.
Heijungs, R., Guinée, J.B., Kleijn, R., Rovers, V., 2007. Bias in normalization: causes, consequences, detection and remedies. Int. J. LCA 12, 211–216.
Heijungs, R., Goedkoop, M., Struijs, J., Effting, J., Sevenster, M., Huppes, G., 2003. Towards a life cycle impact assessment method which comprises category indicators at the midpoint and the endpoint level. Report of the first project phase: design of the new method.
Hellweg, S., Hofstetter, T.B., Hungerbühler, K., 2001. Modeling waste incineration for life cycle inventory analysis in Switzerland. Environ. Model. Assess. 6, 219–235.
Hellweg, S., Doka, G., Finnveden, G., Hungerbühler, K., 2003a. Waste and ecology: which technologies perform best? In: Ludwig, C., Hellweg, S., Stucki, S. (Eds.), Municipal Solid Waste Management. Springer, Berlin, Heidelberg, New York, pp. 350–404.
Hellweg, S., Hofstetter, T.B., Hungerbühler, K., 2003b. Discounting and the environment. Should current impacts be weighted differently than impacts harming future generations? Int. J. LCA 8, 8–18.
Hellweg, S., Demou, E., Scheringer, M., McKone, T., Hungerbuehler, K., 2005. Confronting workplace exposure to chemicals with LCA: the examples of trichloroethylene and tetrachloroethylene in metal-degreasing and dry-cleaning. Environ. Sci. Technol. 39, 7741–7748.
Hellweg, S., Demou, E., Bruzzi, R., Meijer, A., Rosenbaum, R.K., Huijbregts, M.A.J., McKone, T.E., 2009. Integrating indoor air pollutant exposure within life cycle impact assessment. Environ. Sci. Technol. 43, 1670–1679.
Hertwich, E.G., 2005a. Consumption and the rebound effect: an industrial ecology perspective. J. Indust. Ecol. 9 (1–2), 85–98.
Hertwich, E.G., 2005b. Life cycle approaches to sustainable consumption: a critical review. Environ. Sci. Technol. 39, 4673–4684.
Hettelingh, J.-P., Posch, M., Potting, J., 2005. Country-dependent characterization factors for acidification in Europe. A critical evaluation. Int. J. LCA 10, 177–183.
Hofstetter, P., 1998. Perspectives in Life Cycle Impact Assessment – A Structured Approach to Combine Models of the Technosphere, Ecosphere and Valuesphere. Kluwer Academic Publishers, Dordrecht.
Hofstetter, P., Norris, G., 2003. Why and how should we assess occupational health impacts in integrated product policy? Environ. Sci. Technol. 37, 2025–2035.
Höjer, M., Ahlroth, S., Dreborg, K.-H., Ekvall, T., Finnveden, G., Hjelm, O., Hochschorner, E., Nilsson, M., Palm, V., 2008. Scenarios in selected tools for environmental systems analysis. J. Cleaner Prod. 16, 1958–1970.
Huijbregts, A.M.J., Hellweg, S., Frischknecht, R., Hungerbühler, K., Hendriks, A.J., 2008. Ecological footprint accounting life cycle assessment of products. Ecol. Economics 64, 798–807.
Huijbregts, M., Schöpp, W., Verkuijlen, E., Heijungs, R., Reijnders, L., 2000. Spatially explicit characterisation of acidifying and eutrophying air pollution in life-cycle assessment. J. Indust. Ecol. 4 (3), 125–142.
Huijbregts, M.A.J., 1998. Application of uncertainty and variability in LCA. Part I: A general framework for the analysis of uncertainty and variability in life cycle assessment. Int. J. Life Cycle Assess. 3, 273–280.
Huijbregts, M.A.J., Thissen, U., Guinée, J.B., Jager, T., Kalf, D., van de Meent, D., Ragas, A.M.J., Wegener Sleeswijk, A., Reijnders, L., 2000b. Priority assessment of toxic substances in life cycle assessment. Part I: Calculation of toxicity potentials for 181 substances with nested multi-media fate, exposure and effects model USES-LCA. Chemosphere 41, 541–573.
Huijbregts, M.A.J., Norris, G., Bretz, R., Ciroth, A., Maurice, B., von Bahr, B., Weidema, B.P., de Beaufort, A.S.H., 2001. Framework for modelling data uncertainty in life cycle inventories. Int. J. Life Cycle Assess. 6, 127–132.
Huijbregts, M.A.J., Rombouts, L.J.A., Hellweg, S., Frischknecht, R., Dik van de Meent, J.H., Ragas, A.M.J., Reijnders, L., Struijs, J., 2006. Is cumulative fossil energy demand a useful indicator for the environmental performance of products? Environ. Sci. Technol. 40, 641–648.
Hunkeler, D., Rebitzer, G., 2005. The future of life cycle assessment. Int. J. LCA 10, 305–308.
Hunkeler, D., Lichtenvort, K., Rebotzer, G. (Eds.), 2008. Environmental Life Cycle Costing. SETAC, Pensacola, FL.
Huppes, G., de Koning, A., Suh, S., Heijungs, R., van Oers, L., Nielsen, P., Guinée, J.B., 2006. Environmental impacts of consumption in the European union using detailed input–output analysis. J. Indust. Ecol. 10 (3), 129–146.
Huppes, G., Davidson, M.D., Kuyper, J., van Oers, L., Udo de Haes, H.A., Warringa, G., 2007. Eco-efficient environmental policy in oil and gas production in The Netherlands. Ecol. Economics 61, 43–51.
Ibenholt, K., 2002. Materials flow analysis and economic modelling. In: Ayres, R.U., Ayres, L.W. (Eds.), Handbook of Industrial Ecology. Edward Elgar, Cheltenham, pp. 177–184.
IISI, 2007. International Iron and Steel Institute (IISI). LCI data on steel production. <http://www.worldstainless.org/About+stainless/Ss+and+he/LCI/> (data available on request only).
ISO, 2006a. ISO 14040 International Standard. In: Environmental Management – Life Cycle Assessment – Principles and Framework. International Organisation for Standardization, Geneva, Switzerland.
ISO, 2006b. ISO 14044 International Standard. In: Environmental Management – Life Cycle Assessment – Requirements and Guidelines. International Organisation for Standardisation, Geneva, Switzerland.
Itsubo, N., Inaba, A., 2003. A new LCIA method: LIME has been completed. Int. J. LCA 8, 305.
Itsubo, N., Sakagami, M., Washida, T., Kukubo, K., Inaba, A., 2004. Weighting across safeguard objects through the application of conjoint analysis. Int. J. LCA 9, 196–205.
JEMAI, 2007. Japan Environmental Management Association for Industry (JEMAI). JEMAI database. <http://www.jemai.or.jp/english/index.cfm> (database available in Japanese language only).
Jimenez-Gonzales, C., Kim, S., Overcash, M.R., 2000. Methodology for developing gate-to-gate life cycle inventory information. Int. J. LCA 5, 153–159.
Jolliet, O., 2003. Overview of the LCIA work meeting results in Vienna, Tsukuba and Barcelona. Life Cycle Impact Assessment Definition Study: Background Document II. UNEP-SETAC Life Cycle Initiative.
Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G., Rosenbaum, R., 2003a. IMPACT 2002+: a new life cycle impact assessment methodology. Int. J. LCA 8, 324–330.
Jolliet, O., Brent, A., Heijungs, R., Itsubu, N., Milà I Canals, L., Mueller-Wenk, R., Pena, C., Pennington, D., Schenk, R., Stewart, M., Udo de Haes, H., Weidema, B., 2003b. Analysis of Midpoint Categories. In: Life Cycle Impact Assessment Definition Study: Background Document II. UNEP-SETAC Life Cycle Initiative, UNEP, Paris.
Jørgensen, A., Le Bocq, A., Hauschild, M.Z., 2008. Methodologies for Social Life Cycle Assessment – a review. Int. J. Life Cycle Assess. 13, 96–103.
Joshi, S., 1999. Product environmental life-cycle assessment using input–output techniques. J. Indust. Ecol. 3, 95–120.
Kløverpris, J., Wenzel, H., Nielsen, P.H., 2008. Life cycle inventory modelling of land use induced by crop consumption. Part 1: Conceptual analysis and methodological proposal. Int. J. LCA 13, 13–21.
Koellner, T., Scholz, R., 2007. Assessment of the land use impacts on the natural environment. Part 1. An analytical framework for pure land occupation and land use change. Int. J. LCA 12, 16–23.
Koellner, T., Scholz, R., 2008. Assessment of the land use impacts on the natural environment. Part 2. Generic characterization factors for local species diversity in central Europe. Int. J. LCA 13, 32–48.
Koellner, T., Suh, S., Weber, O., Moser, C., Scholz, R.W., 2007. Do conventional and sustainability investment funds differ in their environmental impacts? – A comparison by means of Input–Output Life Cycle Assessment. J. Indust. Ecol. 11 (3), 41–60.
Köhler, A., 2006. Environmental Assessment of Industrial Wastewater Treatment Processes and Waterborne Organic Contaminant Emissions. Ph.D. thesis No. 16367, ETH Zurich.
Koehler, A., 2008. Water use in LCA: managing the planet's freshwater resources. Int. J. Life Cycle Assess 13, 451–455.
Köhler, A., Hellweg, S., Escher, B., Hungerbühler, K., 2006. Organic pollutant removal versus toxicity reduction in industrial wastewater treatment: the example of wastewater from fluorescent whitening agent production. Environ. Sci. Technol. 40, 3395–3401.
Köhler, A., Hellweg, S., Recan, E., Hungerbühler, K., 2007. Input-dependent life-cycle inventory model of industrial wastewater-treatment processes in the chemical sector. Environ. Sci. Technol. 41, 5515–5522.
Kondo, Y., Nakamura, S., 2004. Evaluating alternative life-cycle strategies for electrical appliances by the waste input–output model. Int. J. Life Cycle Assess. 9, 236–246.
Kremer, M., Goldhan, G., Heyde, M., 1998. Waste treatment in product specific life cycle inventories. An approach of material related modelling. Part 1: Incineration. Int. J. LCA 3, 47–55.
Krewitt, W., Mayerhofer, P., Friedrich, A., Trukenmüller, A., Friedrich, R., 1998. Application of the impact pathway analysis in the context of LCA. Int. J LCA 3, 86–94.
Krozer, J., Viz, J.C., 1998. How to get LCA in the right direction. J. Cleaner Prod. 6, 53–61.
Larsen, H.F., Hansen, M.S., Hauschild, M., 2009. Including chemical-related impact categories in LCA on printed matter – does it matter? J. Cleaner Prod. 17, 115–128.
Lave, L.B., Cobras-Flores, E., Hendrickson, C., McMichael, F., 1995. Using input–output analysis to estimate economy wide discharges. Environ. Sci. Technol. 29, 420–426.
Lenzen, M., 2000. Errors in conventional and input–output-based life-cycle inventories. J. Indust. Ecol. 4, 127–148.
Lesage, P., Ekvall, T., Deschenes, L., Samson, R., 2007a. Environmental assessment of brownfield rehabilitation using two different life cycle inventory models: Part I: Methodological approach. Int. J. Life Cycle Assess 12, 391–398.
Lesage, P., Ekvall, T., Deschenes, L., Samson, R., 2007b. Environmental assessment of brownfield rehabilitation using two different life cycle inventory models: Part 2: Case study. Int. J. Life Cycle Assess. 12, 497–513.
Lindeijer, E., Müller-Wenk, R., Steen, B., 2002. Impact assessment of resources and land use. In: Udo de Haes, H.A., Finnveden, G., Goedkoop, M., Hauschild, M., Hertwich, E.G., Hofstetter, P., Jolliet, O., Klöpffer, W., Krewitt, W., Lindeijer, E.W., Müller-Wenk, R., Olsen, S.I., Pennington, D.W., Potting, J., Steen, B. (Eds.), Life-Cycle Impact Assessment: Striving Towards Best Practise. SETAC Press, Pensacola, FL.

Lindfors, L.-G., Christiansen, K., Hoffman, L., Virtanen, Y., Juntilla, V., Hanssen, O.J., Rönning, A., Ekvall, T., Finnveden, G., 1995. Nordic Guidelines on Life-Cycle Assessment. Nord. Nordic Council of Ministers, Copenhagen, Denmark, p. 20.
Lloyd, S.M., Ries, R., 2007. Characterizing, propagating, and analyzing uncertainty in life-cycle assessment: a survey of quantitative approaches. J. Indust. Ecol. 11 (1), 161–179.
Lo, S.-C., Ma, H.-w., Lo, S.-L., 2005. Quantifying and reducing uncertainty in life cycle assessment using the Bayesian Monte Carlo method. Sci. Total Environ. 340, 23–33.
Lundie, S., Ciroth, A., Huppes, G., 2007a. Inventory methods in LCA: towards consistency and improvement – Final Report. UNEP-SETAC Life Cycle Initiative. <http://lcinitiative.unep.fr/includes/file.asp?site=lcinit&file=1DBE10DB-888A-4891–9C52-102966464F8D>.
Lundie, S., Huijbregts, M.A.J., Rowley, H.V., Mohr, N.J., Feitz, A.J., 2007b. Australian characterisation factors and normalisation figures for human toxicity and ecotoxicity. J. Cleaner Prod. 15, 819–832.
Mattsson, N., 1997. Internalizing Technological Development in Energy Systems Models. LicEng thesis, Chalmers University of Technology, Gothenburg, Sweden.
Mattsson, N., Unger, T., Ekvall, T., 2003. Effects of perturbations in a dynamic system – the case of nordic power production. In: Unger, T. (Ed.), Common Energy and Climate Strategies for the Nordic Countries – A Model Analysis. PhD thesis, Chalmers University of Technology, Göteborg, Sweden.
Mauris, G., Lasserre, V., Foulloy, L., 2001. A fuzzy approach for the expression of uncertainty in measurement. Measurement 29, 165–177.
McDougall, F., White, P., Franke, M., Hindle, P., 2001. Integrated Solid Waste Management: A Life Cycle Inventory, second ed. Blackwell Science, Oxford, UK.
McKone, T.E., Hertwich, E.G., 2001. The human toxicity potential and a strategy for evaluating model performance in life cycle assessment. J. LCA 6, 106–109.
Meijer, A., Huijbregts, M., Reijnders, L., 2005a. Human Health damage due to indoor sources of organic compounds and radioactivity in life cycle impact assessment of dwellings – Part 1: Characterization factors. Int. J. Life Cycle Assess. 10, 309–316.
Meijer, A., Huijbregts, M., Reijnders, L., 2005b. Human health damages due to indoor sources of organic compounds and radioctivity in life cycle impact assessment of dwellings – Part 2: Damage sources. Int. J. Life Cycle Assess. 10, 383–392.
Mettier, T.M., Hofstetter, P., 2005. Survey insights into weighting environmental damages. influence of context and group. J. Indust. Ecol. 8, 189–209.
Michelsen, O., 2008. Assessment of land use impact on biodiversity. Proposal of a new methodology exemplified with forestry operations in Norway. Int. J. LCA 13, 22–31.
Mila i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Freiermuth Knuchel, R., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B., 2007a. Key elements in framework for land use impact assessment within LCA. Int. J. LCA 12, 5–15.
Mila i Canals, L., Müller-Wenk, R., Depestele, J., Dubreuil, A., Freiermuth Knuchel, R., Gaillard, G., Michelsen, O., Bauer, C., Rydgren, B., 2007b. Key elements in framework for land use impact assessment within LCA. Response to Helias Udo de Haes. Int. J. LCA 12, 2–4.
Molander, S., Lidholm, P., Schowanek, D., Recasens, M., Fullana, P., Christensen, F.M., Guinée, J.B., Hauschild, M., Jolliet, O., Carlson, R., Pennington, D.W., Bachmann, T.M., 2004. OMNIITOX – operational life-cycle impact assessment models and information tools for practitioners. Int. J. LCA 9, 282–288.
Morgan, M.G., Henrion, M., 1990. Uncertainty. A Guide to Dealing with Uncertainty in Quantitative Risk and Policy Analysis. Cambridge University Press, Cambridge.
Moriguchi, Y., Kondo, Y., Shimizu, H., 1993. Analyzing the life cycle impact of cars: the case of CO_2. Indust. Environ. 16 (1–2), 42–45.
Nansai, K., Moriguchi, Y., Tohno, S., 2002. Embodied Energy and Emission Intensity Data for Japan using Input–Output Tables (3EID)—Inventory Data for LCA Center for Global Environmental Research, Japan.
Nansai, K., Moriguchi, Y., Tohmo, S., 2003. Compilation and application of Japanese inventories for energy consumption and air pollutant emissions using input–output tables. Environ. Sci. Technol. 37, 2005–2015.
Narita, N., Nakahara, Y., Morimoto, M., Aoki, R., Suda, S., 2004. Current LCA database development in Japan – results of the LCA project. Int. J. Life Cycle Assess. 9, 355–359.
Ness, B., Urbel-Piirsalu, E., Anderberg, S., Olsson, L., 2007. Categorising tools for sustainability assessment. Ecol. Economics 60, 498–508.
Nielsen, P.H., Hauschild, M., 1998. Product specific emissions from municipal solid waste landfills; Part I: Landfill model. Int. J. LCA 3, 158–168.
Nilsson, M., Eckerberg, K. (Eds.), 2007. Environmental Policy Integration in Practise. Shaping Institutions for Learning. Earthscan.
Nilsson, M., Björklund, A., Finnveden, G., Johansson, J., 2005. Testing an SEA methodology for the energy sector – a waste incineration tax proposal. Environ. Impact Assess. Rev. 25, 1–32.
NREL, 2004. National Renewable Energy Laboratory (NREL). US Life Cycle Inventory Database. NREL, Golden, CO. <http://www.nrel.gov/lci/>.
Obersteiner, G., Binner, E., Mostbauer, P., Salhofer, S., 2007. Landfill modelling in LCA – a contribution based on empirical data. Waste Manage 27, S58–S74.
Olsen, S.I., Christensen, F.M., Hauschild, M., Pedersen, F., Larsen, H.F., Tørsløv, J., 2001. Life cycle impact assessment and risk assessment of chemicals – a methodological comparison. Environ. Impact Assess. Rev. 21, 385–404.
Oreskes, N., Shrader-Frechette, K., Belitz, K., 1994. Verification, validation, and confirmation of numerical models in the earth sciences. Science 263, 641–645.
Owens, J.W., 2002. Water resources in life-cycle impact assessment: considerations in choosing category indicators. J. Indust. Ecol. 5 (2), 37–54.
Palm, V., Wadeskog, A., Finnveden, G., 2006. Swedish experiences of using environmental accounts data for integrated product policy (IPP) issues. J. Indust. Ecol. 10 (3), 57–72.
Pant, R., Van Hoof, G., Schowanek, D., Feijtel, T.C.J., De Koning, A., Hauschild, M., Olsen, S.I., Pennington, D.W., Rosenbaum, R., 2004. Comparison between three different LCIA methods for aquatic ecotoxicity and a product environmental risk assessment: insights from a detergent case study within OMNIITOX. Int. J. Life Cycle Assess. 9, 295–306.
Patel, M., 2009. Sustainability assessment of technologies – mid term and long term research to improve the sustainability evaluation of technologies <http://www.calcasproject.net/includes/file.asp?site=calcas&file=830DD11F-FEC4–4BA9-8E63-B43FE0F52EF9>.
Pennington, D., Crettaz, P., Tauxe, A., Rhomberg, L., Brand, K., Jolliet, O., 2002. Assessing human health response in life cycle assessment using ED10s and DALYs: Part 2 – Noncancer effects. Risk Anal. 22, 947–963.
Pennington, D.W., Potting, J., Finnveden, G., Lindeijer, E.W., Jolliet, O., Rydberg, T., Rebitzer, G., 2004. Life cycle assessment (Part 2): Current impact assessment practise. Environ. Int. 30, 721–739.
Peters, G.P., Hertwich, E.G., 2006. A comment on "Functions, commodities and environmental impacts in an ecological–economic model". Ecol. Economics 59, 1–6.
Pfister, S., Koehler, A., Hellweg, S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environ. Sci. Technol. 43, 4098–4104.
Potting, J., Hauschild, M.Z., 1997. Predicted environmental impact and expected occurrence of actual environmental impact. Part 2: Spatial differentiation in life-cycle assessment via the site-dependent characterisation of environmental impact from emissions. Int. J. LCA 2, 209–216.
Potting, J., Schöpp, W., Blok, K., Hauschild, M.Z., 1998. Site-dependent life-cycle assessment of acidification. J. Indust. Ecol. 2, 63–87.
Pré Consultants, 2008. Personal Communication. Pré Consultants, The Netherlands.
Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., Schmidt, W.-P., Suh, S., Weidema, B.P., Pennington, D.W., 2004. Life cycle assessment – Part 1: Framework, goal & scope definition, inventory analysis, and applications. Environ. Int. 30, 701–720.
Rebitzer, G., Stoffregen, A., Gerber, J., 2007. Modeling water consumption and water use: a first comprehensive approach to address water resources in LCA. In: Presentation at the SETAC Europe Annual Meeting. Porto, May 20–24, 2007.
RMIT, 2007. Centre for Design, RMIT University. Australian LCI Database, Available at: http://www.auslci.com/.
Rosenbaum, R.K., Bachmann, T.M., Hauschild, M.Z., Huijbregts, M.A.J., Jolliet, O., Juraske, R., Köhler, A., Larsen, H.F., MacLeod, M., Margni, M., McKone, T.E., Payet, J., Schuhmacher, M., Russel, A., van de Meent, D., 2007. USEtox – the UNEP/SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in Life Cycle Impact Assessment. Int. J. Life Cycle Assess. 13, 532–546.
Salhofer, S., Wasserman, G., Binner, E., 2006. Strategic environmental assessment as an approach to assess waste management systems. Experiences from an Austrian case study. Environ. Model. Software 22, 610–618.
Sandén, B., Karlström, M., 2007. Positive and negative feedback in consequential life-cycle assessment. J. Clean. Prod. 15, 1469–1481.
SCB, 2008. Swedish environmental accounts – data and analysis. <http://www.mir.scb.se/Eng_Default.htm> (accessed January 2008).
Schmidt, A., Poulsen, P., Andreasen, J., Floe, T., Poulsen, K., 2004. LCA and the Working Environment Environmental Project 907 2004. Danish EPA, Copenhagen (Denmark).
Seppälä, J., 2007. On the meaning of fuzzy approach and normalisation in life cycle impact assessment. Int. J. Life Cycle Assess. 12, 464–469.
Seppälä, J., Posch, M., Johansson, M., Hettelingh, J.-P., 2006. Country-dependent characterization factors for acidification and terrestrial eutrophication based on accumulated exceedence as an impact category indicator. Int. J. LCA 11, 403–416.
Seyler, C., Hellweg, S., Monteil, M., Hungerbühler, K., 2005a. Life cycle inventory for use of waste solvent as fuel substitute in the cement industry: a multi-input allocation model. Int. J. LCA 10, 120–130.
Seyler, C., Hofstetter, T.B., Hungerbühler, K., 2005b. Life cycle inventory for thermal treatment of waste solvent from chemical industry: a multi-input allocation model. J. Cleaner Prod. 13, 1211–1224.
Soares, S.R., Toffoletto, L., Dechênes, L., 2006. Development of weighting factors in the context of LCIA. J. Cleaner Prod. 14, 649–660.
Solano, E., Ranjithan, S., Barlaz, M.A., Brill, E.D., 2002. Life cycle-based solid waste management – 1. Model development. J. Environ. Eng. 128, 981–992.
Spielmann, M., Scholz, R.W., Tietje, O., de Haan, P., 2005. Scenario modelling in prospective LCA of transport systems. Application of formative scenario analysis. Int. J. Life Cycle Assess. 10, 325–335.
Spielmann, M., de Haan, P., Scholz, R.W., 2008. Environmental rebound effects of high-speed transport technologies: a case study of climate change rebound effects of a future underground maglev train system. J. Cleaner Prod. 16, 1388–1398.
Steen, B., 1999. A Systematic Approach to Environmental Priority Strategies in Product Development (EPS). Version 2000 – General System Characteristics/Models and Data of the Default Method. CPM Report 1999 and CPM Report 1999, Chalmers University of Technology, Gothenburg, Sweden, p. 4 and 5.
Stewart, M., Weidema, B., 2005. A consistent framework for assessing the impacts from resource use. Int. J. LCA 10, 240–247.

Stewart, M., Goedkoop, M., 2003. LCIA Needs Analysis Report of the UNEP-SETAC Life Cycle Initiative. Life Cycle Impact Assessment Definition Study: Background Document I. UNEP-SETAC Life Cycle Initiative.
Stranddorf, H., Hoffmann, L., Schmidt, A., 2005. Impact Categories, Normalisation and Weighting in LCA. Environmental News No. 78, Danish Environmental Protection Agency, Copenhagen.
Suh, S., 2004. Functions, commodities and environmental impacts in an ecological economic model. Ecol. Economics 48, 451–467.
Suh, S., 2005. Developing sectoral environmental database for input–output analysis: comprehensive environmental data archive of the US. Economic Syst. Res. 17, 449–469.
Suh, S., 2006. Reply: downstream cut-offs in integrated hybrid life cycle assessment. Ecol. Economics 59, 7–12.
Suh, S., Huppes, G., 2002. Missing inventory estimation tool using extended input–output analysis. Int. J. Life Cycle Assess. 7, 134–140.
Suh, S., Huppes, G., 2005. Methods for life cycle inventory of a product. J. Cleaner Prod. 13, 687–697.
Suh, S., Lenzen, M., Treloar, G.J., Hondo, H., Horvath, A., Huppes, G., Jolliet, O., Klann, U., Krewitt, W., Moriguchi, Y., Munksgaard, J., Norris, G., 2004. System boundary selection in life-cycle inventories using hybrid approaches. Environ. Sci. Technol. 38, 657–664.
Sundqvist, J.-O., Finnveden, G., Stripple, H., Albertsson, A.-C., Karlsson, S., Berendson, J., Höglund, L.-O., 1997. Life Cycle Assessment and Solid Waste – Stage 2. AFR. Swedish Environmental Protection Agency, Stockholm.
Tan, R., Briones, L.M.A., Culaba, A., 2007. Fuzzy data reconciliation in reacting and non-reacting process data for life cycle inventory analysis. J. Cleaner Prod. 15, 944–949.
Tan, R.R., 2008. Using fuzzy numbers to propagate uncertainty in matrix-based LCI. Int. J. Life Cycle Assess 13, 585–592.
Thiesen, J., Christensen, T.S., Kristensen, T.G., Andersen, R.D., Brunoe, B., Gregersen, T.K., Thrane, M., Weidema, B.P., 2008. Rebound effects of price differences. Int. J. LCA 13, 104–114.
Thomassen, M.A., Dalgaard, R., Heijungs, R., de Boer, I., 2008. Attributional and consequential LCA of milk production. Int. J. Life Cycle Assess. 13, 339–349.
Tillman, A.-M., 2000. Significance of decision-making for LCA methodology. Environ. Impact Assess. Rev. 20, 113–123.
Tillman, A.-M., Ekvall, T., Baumann, H., Rydberg, T., 1994. Choice of system boundaries in life cycle assessment. J. Cleaner Prod. 2, 21–29.
Tiruta-Barna, L., Benetto, E., Perrodin, Y., 2007. Environmental impact and risk assessment of mineral wastes reuse strategies: review and critical analysis of approaches and applications. Resour. Conserv. Recycling 50, 351–379.
Treloar, G.J., Love, P.E.D., Faniran, O.O., Iyer-Raniga, U., 2000. A hybrid life cycle assessment method for construction. Construct. Manage. Economics 18, 5–9.
Tukker, A., Jansen, B., 2006. Environmental impacts of products: a detailed review of studies. J. Indust. Ecol. 10 (3), 159–182.
UBA, 2007. Umweltbundesamt (UBA) (German Environmental Protection Agency). PROBAS Database. <http://www.probas.umweltbundesamt.de/php/index.php> (available in German language only).
Udo de Haes, H.A., 1993. Applications of Life Cycle Assessment: expectations, drawbacks and perspectives. J. Cleaner Prod. 1, 131–137.
Udo de Haes, H.A. (Ed.), 1996. Towards a Methodology for Life Cycle Impact Assessment. SETAC-Europe. Belgium, Brussels.
Udo de Haes, H., 2006. How to approach land use in LCIA, or how to avoid the Cinderella effect? Comments on 'Key elements in a framework for land use impact assessment within LCA'. Int. J. LCA 11, 219–221.
Udo de Haes, H.A., Jolliet, O., Finnveden, G., Hauschild, M.Z., Krewitt, W., Mueller-Wenk, R., 1999. Best available practice regarding impact categories and category indicators in life cycle impact assessment/background document for the second working group on life cycle impact assessment of SETAC Europe (WIA-2). Int. J. LCA 4, 66–74 and 4, 167–174.
Udo de Haes, H.A., Finnveden, G., Goedkoop, M., Hauschild, M., Hertwich, E.G., Hofstetter, P., Jolliet, O., Klöpffer, W., Krewitt, W., Lindeijer, E.W., Müller-Wenk, R., Olsen, S.I., Pennington, D.W., Potting, J., Steen, B. (Eds.), 2002. Life-Cycle Impact Assessment: Striving Towards Best Practise. SETAC Press, Pensacola, FL.
UK Environment Agency, 2008. http://www.environment-agency.gov.uk/wtd/1396237/?version = 1&lang = _e.
United Nations Environment Programme, 2002. Life Cycle Initiative homepage. <http://www.uneptie.org/pc/sustain/lcinitiative/home.htm>.
United Nations, 1992. Report of the United Nations Conference on Environment and Development, Rio de Janeiro, 3–14 June 1992. Annex I: Rio Declaration on Environment and Development, A/CONF.151/26 (vol. I), 12 August 1992.
Van Zelm, R., Huijbregts, M.A.J., den Hollander, H.A., van Jaarsveld, H.A., Sauter, F.J., Struijs, J., van Wijnen, H.J., van de Meent, D., 2008. European characterization factors for human health damage PM10 and ozone in life cycle impact assessment. Atmos. Environ. 42, 441–453.
Wal-Mart, 2007. S-Mart. Newsletter. <http://www.walmartstores.com/Sustainability> April/May 2007.
Wal-Mart, 2008. S-Mart. Newsletter. <http://www.walmartstores.com/Sustainability> March 2008.
Walz, R., Herrchen, M., Keller, D., Stahl, B., 1996. Impact category ecotoxicity and valuation procedure, ecotoxicological impact assessment and the valuation step within LCA – pragmatic approaches. Int. J. LCA 1, 193–198.
Wegener Sleeswijk, A., Kleijn, R., Meeusen-van Onna, M.J.G., Leneman, H., Sengers, H.H.W.J.M., van Zeijts, H., Reus, J.A.W.A., 1996. Application of LCA to Agricultural Products; 1. Core Methodological Issues; 2. Supplement to the LCA Guide; 3. Methodological Background. Centre of Environmental Science, Leiden University, Leiden.
Wegener Sleeswijk*, A., van Oers, L.F.C.M., Guinée, J.B., Struijs, J., Huijbregts, M.A.J., 2008. Normalisation in product life cycle assessment: an LCA of the global and European economic systems in the year 2000. Sci. Total Environ. 390, 227–240.
Weidema, B., Notten, P., Suh, S., 2006. Prioritising within the product-oriented environmental policy – The Danish perspectives. J. Indust. Ecol. 10 (3), 73–87.
Weidema, B.P., 2003. Market Information in Life Cycle Assessment. In: Environmental Project No. 863. Danish Environmental Protection Agency, Copenhagen.
Weidema, B.P., 2009. Using the budget constraint to monetarise impact assessment results. Ecol. Economics 68, 1591–1598.
Weidema, B.P., Frees, N., Nielsen, A.-M., 1999. Marginal production technologies for life cycle inventories. Int. J. Life Cycle Assess. 4, 448–456.
Weidema, B.P., Ekvall, T., Pesonen, H.-L., Rebitzer, G., Sonneman, G.W., Spielmann, M., 2004. Scenarios in LCA. SETAC, Brussels.
Weidema, B.P., Finnveden, G., Stewart, M., 2005. Impacts from resource use – a common position paper. Int. J. LCA 10, 382.
Weidema, B.P., Wesnæs, M., Hermansen, J., Kristensen, T., Halberg, N., Eder, P., Delgado, L., 2008. Environmental improvement potentials of meat and dairy products. Institute for Prospective Technological Studies, Sevilla. <http://ftp.jrc.es/EURdoc/JRC46650.pdf>.
Wenzel, H., Hauschild, M.Z., Alting, L., 1997. Environmental Assessment of Products. Methodology, Tools, Techniques and Case Studies, vol. 1. Chapman & Hall, United Kingdom, Kluwer Academic Publishers, Hingham, MA, USA, 544 pp. ISBN 0 412 80800 5.
Wernet, G., Hellweg, S., Fischer, U., Papadokonstantakis, S., Hungerbühler, K., 2008. Molecular structure-based models of chemical inventories using neural networks. Environ. Sci. Technol. 42, 6717–6722.
Wright, T.P., 1936. Factors affecting the costs of airplanes. J. Aeronaut. Sci. 3, 122–128.
Zhang, Z., Wu, X., Yang, X., Zhu, Y., 2006. BEPAS – a life cycle building environmental performance assessment system. Build. Environ. 41, 669–675.
Zimmermann, P., Doka, G., Huber, F., Labhardt, A., Menard, M., 1996. Ökoinventare von Entsorgungsprozessen, Grundlagen zur Integration der Entsorgung in Ökobilanzen. Institut für Energietechnik, ETH Zurich. ENET.