Aalborg Universitet

AALBORG UNIVERSITY
DENMARK

Comments on the development of harmonized method for Sustainability Assessment of Technologies (SAT)

Weidema, Bo Pedersen; Thrane, Mikkel

Publication date:
2007

Document Version
Publisher's PDF, also known as Version of record

Comments on the development of harmonized method for Sustainability Assessment of Technologies (SAT)

By Bo Weidema[1,2] and Mikkel Thrane[1],

2007-06-04

[1] Associate professor, The LCA Research group at Aalborg University, Denmark
[2] 2.-0 LCA consultants

Introduction

We welcome the opportunity to provide these comments on the main issues raised in the Background paper to the first workshop on Sustainability Assessment of Technologies (Galatola 2007a) and the subsequent presentation at the SETAC conference (Galatola 2007b).

General comment regarding quantitative versus qualitative approaches

Lacking data and uncertainty is often used as an argument for using a qualitative approach in either inventory and/or impact assessment. However, this argument does not withstand a closer examination, since lacking data and uncertainty can be treated much more explicitly and consistently in a quantitative approach. Even very large uncertainties such as those involved in climate change and toxicity are not an argument for not seeking quantification, nor should they be for issues such as rebound effects, site-specific impacts, forecasting or social impacts. Even when quantification can only establish an order of magnitude, this still allows distinguishing the relative importance of different mechanisms, substances or modes of intervention. Quantitative uncertainty ranges allows to express the current state of knowledge, and can be applied even for cognitive aspects of uncertainty, as when qualitative assessments of data quality are converted into uncertainty ranges (Weidema & Wesnaes 1996), and applied to provide an uncertainty-based data collection strategy (Weidema et al. 2003).

In a qualitative approach, it is very difficult to distinguish between less and more important items. Even when rating on a scale from 0 to 10 or 0 to 100, there is a tendency that all items under consideration are seen as equally important. The reason for this is the psychological mechanisms known as cognitive biases, e.g. ‘anchoring’ (Tversky & Kahneman 1974), which implies that all of the items under consideration are seen as being of some importance, and depending on the scale presented, a person will seek to accommodate all categories on this scale, in not too extreme positions. This means that the results depend very much on what items are at all considered, and at what level of aggregation, and also on the scale and accompanying information.

Qualitative approaches such as the MECO method and the ‘traffic light’ or ‘flagging’ methods may indeed be useful in an initial screening LCA, to identify the areas where it is most important to obtain better (quantitative) data, but should not be seen as a final result or as an excuse for indolence, as when lacking data are set to zero without further justification. Rather, there should be a requirement to investigate and quantify all red (and maybe yellow) flags. Since there is no way to judge the relative importance of different flags, we have often found that when later quantifying the concerned impacts, a yellow flag may appear as more important than a red, while another yellow flag may turn out to represent an insignificant impact.

In the following comments, we will come back to this issue in the context of rebound effects, site-specific impacts, forecasting, and social impacts.

How to define the functional unit

The concept of a functional unit is also applicable to technology assessments, but obviously the scale of the functional unit should not be chosen arbitrarily as is often seen when LCA is applied to small changes in individual products. For a large change, which affects the determining parameters for the overall technology development, it is misleading if the functional unit is chosen independently of the actual scale of the studied substitution. When studying substitutions involving the entire market of a major product or process, e.g. studies dealing with regional or countrywide waste handling systems, organic agriculture, renewable energy systems, or studies dealing with legislation or standards for an entire industry, it is relevant to choose a functional unit of the same size as the affected market (Weidema et al. 2004).

The functional unit should be broad enough to include the product alternatives, which can substitute each other in the relevant market. A more narrow selection will give a result that does not reflect the true potential for environmental improvement, while a broader selection would include irrelevant alternatives (Weidema et al. 2004). For example, if the function is seen as 'an automobile in its average application', then this obviously limits the improvement options compared to viewing the product as 'a means of passenger transport' or even wider as 'a means of co-locating a person with a desired object, activity or (group of) person(s)', which allows to include non-transport alternatives, such as telecommunication, to fulfil the co-location need.

The broad definition of the functional unit also allows for the inclusion of rebound effects. Rebound effects are the derived changes in production and consumption when the implementation of a technology liberates or binds a scarce production or consumption factor, such as:

- money (when the new technology is more or less costly than the current technology),
- time (when the new technology is more or less time consuming than the current technology)
- space (when the new technology takes up more or less space than the current technology), or
- technological availability (when the new technology affects the availability of other specific technologies or production factors, e.g. raw materials).

We may distinguish three types of rebound effects:

- Specific rebound effects, where production and consumption of the specific product in question changes.
- General rebound effects, where the overall production and consumption changes.
- Behavioural rebound effects, where the organisation of production and consumption changes, affecting both the specific product in question and other specific products.

Price rebounds occur when the new technology is more or less costly than the current technology. The price rebound may be divided in an effect on the consumption of the specific product in question (the specific price rebound), and an effect on the general consumer expenditure (the general price rebound, also known as the income effect) distinguished by the own-price elasticity and general price elasticity. For example, a own-price elasticity of 0.35 implies that the price rebound can be quantified as a 35% change in the consumption of the product in question and a 65% change in average consumption (or rather the marginal household consumption as determined by Thiesen et al. 2006).

Time rebounds occur when the new technology is more or less time consuming than the current technology. This is particularly relevant for changes in behaviour, e.g. when the home delivery of groceries will liberate time previously spent for shopping. While such time-savings are easily quantified, the required time elasticities, i.e. coefficients of time allocation between different activities when more or less time becomes available, are unfortunately scarce. Gershuny (2002) reports time elasticities based on UK time series, covering the situations where the saved time is released from unpaid work. However, own-time elasticities *within* unpaid work may be much larger, e.g. when the time saved on shopping is instead used to increase the time spent for child care or cooking. Most activities within unpaid work, apart from shopping, do not involve transport or other activities with high environmental intensities. Time rebounds that stay within unpaid work may therefore often be environmentally neutral. Better data on time elasticities would nevertheless be desirable.

In addition to the time spent on different activities, there may also be an additional rebound effect from shifting in the timing of activities. The shifting of an activity in time may influence the overall activity pattern, even when there is no change in overall time spent on the specific activity. For example, while shopping is normally done in the day-time, Internet-shopping may be done e.g. at night-time. The liberated day-time may have a different alternative use than the additional time used at night. In general, improvements that provide more flexible time usage, such as Internet shopping, are likely to release more time for out-going activities requiring transport, which accentuates the possibility that the environmental improvement is partly offset by the rebound effect.

Space rebounds occur when the new technology takes up more or less space than the current technology, e.g. when involving changes in agricultural land use or road space. Quantification of these rebound effects are also possible, see for example the review article in TDM Encyclopedia (Victoria Transport Policy Institute 2007) documenting that a technology that liberates road space will reduce congestion and subsequently induce increased traffic which in the long-term will fill a significant portion (50-90%) of the additional road space released, thus reducing the environmental improvement proportionally.

Technology rebounds occur when the new technology affects the availability of other specific technologies or raw materials. Many new technologies have wider applications than originally foreseen, and the effects may therefore be larger than when looking at the more narrow intended or targeted application. Also, a new technology may reduce the demand for other existing technologies, as for example when home delivery of groceries may lead to decreased car-ownership for families where the need for a car for shopping is a determining factor for car-ownership.

As can be seen from the above examples, most rebound effects are readily quantifiable and our experience is that the impacts related to rebound effects (including marginal spending) can be very substantial and in some cases even larger than the impact of the product, technology or change studied. If treated qualitatively and outside the LCA model, the rebound effects may easily be forgotten or neglected.

The issue of breadth of the functional unit and the options to include rebound effects is also related to what Galatola (2007a) calls level-2 assessments, where it becomes imperative to account for autonomous developments in the background scenarios, including interactions and feedback loops between different technologies. This is very similar to the modelling of rebound effects, and is likewise facilitated or even automatically induced by an adequately broad definition of the functional unit.

Choice of specific cut-off criteria

The use of cut-off criteria has been abandoned in modern LCAs due to the much improved data availability from hybrid databases (databases integrating input-output data from national statistics). Since background data are available on all activities in society, there is simply no longer any need to exclude specific activities from the assessments. This has led to a significant increase in the consistency and completeness of LCA results, and has made the discussion on cut-off criteria redundant.

Which data to use

We strongly support the idea to use a common European reference database, such as the ELCD, to increase the inter-comparability of LCAs and to reduce the costly data collection efforts. We therefore find it of utmost importance that the ELCD database becomes a high-quality, reliable and complete database. For this reason, we have made the following recommendations to JRC regarding the development of the ELCD database (Weidema et al. 2006):

- The database should apply a hybrid technique combining input-output data and process-based data, in order to eliminate data gaps in the latter.
- The applied input-output data should be disaggregated as much as possible, by using the underlying micro-data and/or with the aid of process-based data, in order that the resulting database obtains the largest degree of detail and validity.
- Improvements should be made in the collection of micro-data for input-output matrices, so that the resulting matrices are better suited for environmental analyses. Especially with respect to material recycling, a better distinction should be achieved between primary and secondary materials production.
- System expansion should be consistently applied to avoid allocation for processes with co-products.

It is our understanding that the experts participating in the first workshop have not been adequately aware of the significant potential of hybrid databases, which combine the completeness of monetary or physical input-output data with the detail of process-based data. The significant gaps that can be found in process-based data have been documented by comparing with corresponding input-output data. Most of the more recent comparisons are available only in the grey literature, but some older published examples are Mongelli et al. (2005) and Treloar (2004). A more theoretical expose is given by Lenzen (2001). As shown by Suh et al. (2004) selecting a system boundary in compliance with ISO standards is, in practice, impossible without using input-output data, and hybrid techniques using therefore form a central element of ISO-compatible system boundary selection practices. The two data sources can readily be integrated and used for cross-validation and should therefore be seen as complementary rather than competing (Weidema 2003). In relation to dynamic modelling, scenario building and modelling of rebound effects, it should also be noted that input-output based databases lend themselves more readily to consistent modifications on an aggregated level.

To be applicable for SAT, a number of reference background scenarios at 5-year intervals should also be included in the ELCD database, i.e. each process should be available in a number of scenario versions for each 5-year period. The specific scenarios for each SAT may then be superimposed on these reference background scenarios.

Which impacts to assess
In principle, all impacts that can be quantified by an LCA can also be quantified for SAT.

It is correct that when a future activity cannot be located exactly either in time or in geographical location, this increases the uncertainty in assessing its impacts for most impact categories. This variation in impacts is a true reflection of the uncertainty of not knowing the location of a future activity, and is not an argument for leaving the impacts unquantified. Rather, it is an argument for including the full range of possible future locations in the assessment and for recording the resulting uncertainty on the central value.

Likewise, the uncertainty related to occurrence of accidents should not be used as an argument for leaving the risk of accidents unquantified. The central value can be determined as the probability of occurrence multiplied by the impact of an average incident. The uncertainty around this value can be determined from the uncertainty on both the probability and the average impact.

Technology Category Rules
Taking over the PCR concept from EPD's may be an unnecessary straitjacket for SAT.

While it is true that different technologies have their own peculiarities that may need specific inventory data or impact categories, the difference between technologies should not be exaggerated. All technologies have the same fundamental system elements and it could lead to inconsistencies if the rules for their assessment were made too technology-specific. Some technologies are cross-cutting and affect many different product systems or other technologies, and it may in practice be very difficult to identify to which technology family a specific new technology belongs – it may in fact belong to more than one…

Another aspect from EPD's that is suggested for SAT is the division of the results on life cycle stages. However, even for EPD's this division is difficult to make in a consistent way, and it is questionable what the purpose of the division is, what message it seeks to convey, and how this message is applied in the decision-making context.

The economic dimension
There seems to be a misconception about Life Cycle Costing (LCC) as a 'third leg' of sustainability. The concept of 'three pillars of sustainability' (human well-being, natural life-support and economic growth, often summarized by the catchy phase 'people, planet, profit') coined by the World Business Council for Sustainable Development has become a widespread 'operational' definition of sustainability. The three pillars provide a division of the environment (defined in ISO 14004 as the surroundings of an organisation), which is useful because of its completeness: An impact is either affecting instrumental values ('profit' or resource productivity) or inherent values, and the inherent values can then again be divided in those related to human well-being ('people') and to the well-being of the non-human environment ('planet'). Some impacts are captured (internalised) in the price of a product (in economic terms known as private costs), while others (known as externalities) are borne by persons outside the product system and not compensated. All impacts on 'people, planet, or profit' are either private costs or externalities, as shown in the table below:

	People	Planet	Profit
Private costs	E.g. health and safety expenditures, product liability expenses	E.g. costs of pollution prevention	Costs of raw materials, wages, taxes, interest on capital
Externalities	E.g. reduction in human well-being due to pollution	E.g. biodiversity impacts from pollution	E.g. reduction in productivity due to human health impacts, missing education due to child labour, or reduction in crop yields due to pollution

In the tentative SETAC definition, LCC only covers the *private costs* of the product over its lifetime. This is normally equal to the economic value over its lifetime, i.e. what the user is willing to pay for the product, and which is therefore covered by its lifetime price. Traditionally, LCA covers only *some of the externalities*, notably (some of) the health aspects of human well-being (i.e. not the social dimension, see next section) and (some of) the impacts on nature (Planet). Traditionally, LCA has ignored impacts on resource productivity (the bottom, right hand box in the table), with the exception of attempts to include an indicator of natural resource depletion.

The misconception about LCC is now that it should cover the third ‘Profit’ leg of sustainability when in fact it simply covers the private cost part of all three legs. Thus, a true sustainability assessment shall also include the ‘Profit’ externalities currently not covered by traditional LCA, i.e. all the impacts on resource productivity, such as the social costs of corruption and perverse subsidies, and the wages lost due to health impacts, missing education, unequal treatment and unemployment, while also including the excess value (i.e. the value in excess of the private costs) of contributions to physical and social infrastructure. Also the ‘People’ and ‘Planet’ part of LCA impact assessment could benefit from a ‘completeness check’ based on current epidemiological knowledge about what are the large impacts and their causes.

The social dimension

Fundamentally, there is no reason that a LC Social Assessment should deviate from ISO14040 (Weidema 2002, 2005). As for the biophysical impacts you can:

- have a company perspective and/or a societal perspective on your impacts,
- rely on one common reference database quantifying the average impacts per process,
- operate with the same system description, with the same allocation procedures,
- describe the impact pathways for each social inventory indicator through midpoints to endpoints, expressed in e.g. QALY – see for instance Weidema (2006a)

Weidema (2006b) show how all existing social inventory indicators (including those from the OECD guidelines for Multinational Enterprises, the Global Reporting Initiative, SA 8000, etc.) could be summarised into a list of 30 mutually independent social inventory indicators, of which 14 are relevant for industrial activities (the remaining 16 being relevant for analyses of governmental activities only). For these 14 inventory indicators, Weidema (2006b) suggests quantitative measurement units, data sources, default reference data, as well as tentative impact pathway descriptions, characterisation factors and normalisation values in line with ISO 14040.

In addition to the quantitative social assessment, we recommend a procedural approach to ensure stakeholder involvement and empowerment. The basis of this should be a power analysis, identifying the affected parties, their interests and power relations, as well as a quantitative assessment of the distribution of the identified impacts upon the identified societal and/or stakeholder groups. Based on this, a participatory procedure involving representatives of the affected parties, empowered to the extent necessary to ensure a fair and free negotiation, should aim at a result that is acceptable for all stakeholders, possibly via an agreed compensation mechanism. An important element in such a procedure could be the Consensus Conference developed by the Danish Board of Technology[1] and already successfully applied in technology assessment.

As already mentioned by Johannes Kreißig (2007), a technology must always be seen in relation to its application. Nevertheless, most of the descriptions of SAT appear to apply a quite narrow technical definition of technology that focus on the 'hardware' dimension only. Other important aspects of a technology are the 'organizational' dimension, as well as the 'knowledge' dimension – not to mention the 'product' that is the point of departure for life cycle thinking. We would therefore encourage the Commission to apply a broader technology concept, which include at least four dimensions: technique, organization, knowledge and product – in accordance with the definition applied in Lorentzen (1988).

Conclusion: In what way does SAT place increased methodological demands on LCA?
In general, there are no restrictions in applying LCA to SAT. However, this application emphasises some key points of LCA that may need to be reinforced, especially a more stringent understanding of the functional unit in terms of scale and breadth, a more systematic approach to including rebound effects, a more standardised procedure for scenario development, an explicit treatment of quantitative uncertainty, and a more complete inclusion of important impact categories. All of these aspects are already in development as part of the LCA methodology, but could indeed be strengthened by forcing the LCA methodology on the more demanding application area presented by SAT.

References
Galatola M. (2007a). Background paper. Pp. 10-17 in EC: "Sustainability Assessment of Technologies (SAT)". Report from Workshop, Brussels, 2007.04.24-25.
Galatola M. (2007b). Towards a European Methodology for Sustainability Assessment of Technologies (SAT). Presentation to the SETAC Europe 17th Annual Meeting, Porto, 2007.05.20-24.
Gershuny J. (2002). Web-use and Net-nerds: A Neo-Functionalist Analysis of the Impact of Information Technology in the Home. ISER Working Paper 2002-1. Colchester: University of Essex. (cited from Hofstetter & Madjar 2003).
Hofstetter P, Madjar M. (2003). Linking change in happiness, time-use, sustainable consumption, and environmental impacts; An attempt to understand time-rebound effects. Zürich: Büro für Analyse und Oekologie.
Kreißig J. (2007). Sustainability Assessment of the Technology family "Housing". Pp. 133-145 in EC: "Sustainability Assessment of Technologies (SAT)". Report from Workshop, Brussels, 2007.04.24-25.

[1] <http://www.tekno.dk/subpage.php3?article=468&toppic=kategori12&language=uk>

Lenzen M. (2001). Errors in conventional and Input-Output-based life-cycle inventories. Journal of Industrial Ecology 4(4):127-148.
Lorentzen A. (1988). Technological Capacity – a contribution to a comprehensive understanding of technology and development in an international perspective. Aalborg University Press. Technology and Society Series no. 5.
Mongelli I, Sangwon Suh S, Huppes G. (2005). A Structure Comparison of two Approaches to LCA Inventory Data, Based on the MIET and ETH Databases. International Journal of Life Cycle Assessment 10(5):317-324.
Suh S, Lenzen M, Treloar G, Hondo H, Horvath A, Huppes G, Jolliet O, Klann U, Krewitt W, Moriguchi Y, Munksgaard J, Norris G A. (2004). System boundary selection for life cycle inventories. Environmental Science & Technology 38(3):657-664. <http://dx.doi.org/10.1021/es0263745>
Thiesen J, Christensen T S, Kristensen T G, Andersen R D, Brunoe B, Gregersen T K, Thrane M, Weidema B P. (2006). Rebound Effects of Price Differences. International Journal of Life Cycle Assessment Online First. <http://www.scientificjournals.com/sj/lca/Abstract/ArtikelId/8719>
Treloar G. (2004). Hybrid Life-Cycle Inventory for Road Construction and Use. Journal of Construction Engineering and Management 130(1):43-49.
Tversky A, Kahneman D. (1974). Judgment under uncertainty: Heuristics and biases. Science 185:1124-1130
Victoria Transport Policy Institute (2007). Rebound Effects. Implications for Transport Planning. TDM Encyclopedia. <http://www.vtpi.org/tdm/tdm64.htm>.
Weidema B P. (2002). Quantifying Corporate Social Responsibility in the value chain. Presentation for the Life Cycle Management Workshop of the UNEP/SETAC Life Cycle Initiative at the ISO TC207 meeting, Johannesburg, 2002-06-12. <http://www.lca-net.com/files/csr.pdf>
Weidema B P. (2003). Matching bottom-up and top-down for verification and integration of LCI databases. Presentation to the International Workshop on LCI-Quality, Karlsruhe, 2003.10.20-21. <http://www.lca-net.com/publications/matching/>
Weidema B P. (2005). ISO 14044 also applies to social LCA. Letter to the editor. International Journal of Life Cycle Assessment 10(6):381. <http://www.lca-net.com/files/social_LCA.pdf>
Weidema B P. (2006a). The Integration of Economic and Social Aspects in Life Cycle Impact Assessment. International Journal of Life Cycle Assessment 11(1):89-96.
Weidema B P. (2006b). Social impact categories, indicators, characterisation and damage modelling. Presentation for the Swiss LCA forum “Life Cycle Perspective for Social Impacts”, Lausanne, 2006.06.15.
Weidema B P, Petersen E H, Øllgaard H, Frees N. (2003). *Reducing uncertainty in LCI.* Copenhagen: Danish Environmental Protection Agency. (Environmental project no. 862). <http://www2.mst.dk/Udgiv/Publications/2003/87-7972-989-4/pdf/87-7972-990-8.PDF>
Weidema B P, Wenzel H, Petersen C, Hansen K. (2004). The product, functional unit and reference flows in LCA. København: Miljøstyrelsen. (Environmental News 70). <http://www2.mst.dk/Udgiv/Publications/2004/87-7614-233-7/pdf/87-7614-234-5.PDF>
Weidema B P, Wesnæs M S. (1996). Data quality management for life cycle inventories - an example of using data quality indicators. Journal of Cleaner Production 4(3-4):167-174.
Weidema B P, Wesnæs M, Christiansen K. (2006). Environmental assessment of municipal waste management scenarios in Malta and Krakow. Report for DG-JRC, Institute for Environment and Sustainability, Soil & Waste Unit, Ispra. Unpublished.