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What life-cycle assessment does and does not do in assessments of waste management

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Abstract

In assessments of the environmental impacts of waste management, life-cycle assessment (LCA) helps expanding the perspective beyond the waste management system. This is important, since the indirect environmental impacts caused by surrounding systems, such as energy and material production, often override the direct impacts of the waste management system itself. However, the applicability of LCA for waste management planning and policy-making is restricted by certain limitations, some of which are characteristics inherent to LCA methodology as such, and some of which are relevant specifically in the context of waste management. Several of them are relevant also for other types of systems analysis. We have identified and discussed such characteristics with regard to how they may restrict the applicability of LCA in the context of waste management. Efforts to improve LCA with regard to these aspects are also described. We also identify what other tools are available for investigating issues that cannot be adequately dealt with by traditional LCA models, and discuss whether LCA methodology should be expanded rather than complemented by other tools to increase its scope and applicability.
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1. Introduction

1.1. Background

Waste management is a complex phenomenon with a range of consequences for the involved stakeholders and the society. One of the many parameters to evaluate is the environmental impact of different treatment options or technical solutions. There are many tools for assessment of environmental impact, but one of the most commonly used is life-cycle assessment (LCA). It helps expanding the perspective beyond the waste management system. This is important since the environmental consequences of waste management often depend more on the impacts on surrounding systems than on the emissions from the waste management system itself (Ekvall, 1999). In particular, the broad perspective of LCA makes it possible to take into account the significant environmental benefits that can be obtained through different waste management processes:

- waste incineration with energy recovery reduces the need for other energy sources,
- material from recycling processes replaces production of virgin material,
- biological treatment may reduce the need for production of artificial fertilisers and vehicle fuel[1],

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[1] It may also help improving the quality of soils, but this is difficult to take into account in LCA.

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- residues from waste incineration may replace gravel at road constructions (Birgisdottir, 2004), etc.

The broad system perspective makes LCA a powerful tool for environmental comparison of different options for waste management of a specific product, a material, or a complex waste flow. Because of this, LCA has gained in acceptance as a tool for waste management planning and policy-making. It is now being used in various contexts, ranging from local planning to policy making at national and international levels. An example of this is the recent thematic strategy on waste management presented by the European Commission.

An international standard for LCA has been developed, and handbooks are available (e.g., Guinée, 2002), as well as scientific reviews of recent developments (Rebitzer et al., 2004; Pennington et al., 2004). Separate publications describe how to apply the method on waste management systems (Finnveden, 1999; Clift et al., 2000). However, to be able to make sustainable use of LCA in the waste management, it is important to be aware of the limitations of the methodology and to understand that the environmental information it generates is neither complete, nor absolutely objective or accurate. The international standardisation process helps to reduce what can appear to be arbitrariness of the methodology, but important methodological choices still remain free to be made in each separate study. The LCA results therefore depend on methodological decisions, for example:

- choice of time perspective (Finnveden et al., 1995; Obersteiner et al., 2007),
- assumptions made in the study,
- sources of input data,
- allocation of environmental burdens to different life cycles (Ekvall and Tillman, 1997; Winkler, 2007), and
- modelling of environmental impacts.

These methodological choices may be influenced by the values and perspectives of the LCA practitioner and the LCA commissioner. This means that an LCA typically does not yield objective answers. The methodology also suffers from large uncertainties (Huijbregts, 1998a,b). As indicated by the references above, the subjective and uncertain aspects of the answers given by LCA have been thoroughly discussed elsewhere. These limitations are also not unique to LCA. Several methods for environmental systems analysis have been developed to support different types of decisions (Wrisberg et al., 2002; Finnveden and Moberg, 2005). Similar problems occur in most of them.

A limitation that has not been much discussed, however, is the fact that a traditional LCA model has several inherent characteristics that prohibit it from giving adequate answers to many significant questions. This is the focus of our paper.

1.2. Aim of the paper

In order to contribute to the awareness of the limitations of LCA, the aim of this paper is to discuss the restrictions in the applicability of LCA as a decision-support tool in waste management planning and policy-making. We do this by identifying certain characteristics of LCA, discuss how these may restrict the applicability of LCA, efforts made to improve LCA methodology with regard to these characteristics, and what other tools are available that cover issues currently not adequately dealt with in LCA. We also discuss whether LCA methodology should be expanded rather than complemented by other tools to increase its scope and applicability. Most of the discussion is valid also for LCA applied outside the waste management sector, and to a large extent it is also valid for other tools for environmental systems analysis.

The advantages and disadvantages of LCA applied to waste management can be discussed at three conceptual levels. The discussion can focus on the characteristics of LCA as a scientific method, on methodological applications of LCA in computer models or methodological guidelines, or on the practical applications of LCA in actual case studies. Our discussion aims at the most general level. The purpose is to shed light on the characteristics of LCA as a scientific method. However, we use examples of methodological applications as well as practical applications as illustrations.

2. Functional unit and system dynamics

2.1. Restrictions in applicability

LCA models of waste management often calculate the environmental burdens per kg or tonne of waste generated. It implies that the quantity of waste is unaffected by the management measures investigated. Having identical amounts of waste treated in different scenarios makes it possible to simplify comparative analyses by neglecting the production and use of the materials (Finnveden, 1999). This simplification is sometimes called the "zero burden assumption", suggesting that the waste carriers none of the upstream burdens into the waste-management system.

LCA models that calculate the environmental burdens per kg or tonne of waste generated allow for environmental comparisons of different options for dealing with this waste, but not for analyses of changes in the quantities of waste generated. They are inadequate for the identification and assessment of waste prevention strategies. They also fail to account for the serious challenges posed by a continuation of the short-term and long-term trends of increasing waste flows, and consequently do not give information on how large capacity for waste treatment is required.

Traditional LCA models are also static. In the context of waste management, this implies that they cannot give information about the appropriate time for investments in waste management plants.

Perhaps more seriously, the system structure and the input data in a traditional LCA both reflect the recent past. This means that, at the best, traditional LCA provides basis for identifying what waste management strategies are best served to solve the needs of the current society. But waste management plants are large investments that will be used for several decades, and the surrounding society can change significantly during this time. A technology that is appropriate today might be incompatible with the long-term sustainability of the society.

2.2. Amendments

A first step towards amending the restrictions imposed by a static assumption with regard to the waste quantity, is to relate the study not only to the composition of the waste but also to the waste quantity (Coleman et al., 2003). This can be made by changing the functional unit to the annual quantity of waste generated in a geographical area. As an example, Xará (2004) used the annual quantity of waste in the city of Porto as the functional unit. Matsui (2004) presented a comparative LCA including different waste management options, as well as waste prevention. The functional unit was a ton of waste generated for the waste management options, and a ton of waste prevented for the waste prevention. Olofsson et al. (2004) also compared waste prevention to different waste management strategies, using the annual quantity of waste in Sweden as the functional unit. The quantity of waste varied between the scenarios because the analysis accounted for the reduction in waste quantity resulting for potential waste prevention measures.

Adjusting the functional unit is obviously a measure to facilitate the assessment of waste prevention. This measure may appear simple enough, but if different scenarios include different waste quantities, the zero burdens assumption is no longer valid. It is reasonable to demand that such studies include the environmental burdens associated with the production of all the materials that eventually become waste. This makes the assessment more complicated.

To be able to plan for changes in waste flows, and to decide on the size of investments in waste-treatment technologies, decision-makers require futures studies of the waste flows. Futures studies include forecasting through, for example, extrapolation and dynamic modelling (Börjeson et al., 2006). Dynamic modelling can also be used for identifying and assessing the efficiency of different strategies for waste prevention. Futures studies also include backcasting, which can be effective for finding routes to a desired, future waste-management system. The different methods for futures studies can be used for defining future waste scenarios that specify the future waste quantities and technologies for waste treatment. LCA is then applied to assess the environmental impact of these scenarios.

However, methods for futures studies can also be integrated in the LCA methodology (Weidema et al., 2004). An example is Olofsson et al. (2004), who made a forecast of the Swedish waste quantity in 2008–2012 as a base case scenario to which the waste prevention was compared. Trisyanti (2004) who performed a systems analysis of solid waste management in Jakarta, also considered the difference in the quantity of waste between 2003 and a forecast for 2015. Björklund and Finnveden (2007) used extrapolated values of Swedish waste quantities to assess the expected effectiveness of a proposed waste incineration tax. When methods for futures studies are integrated in the LCA, the methodology not only assesses scenarios but also assists in developing the scenarios that are to be assessed.

To provide information on the appropriate time for investments, the futures studies probably need to include dynamic models spanning over multiple years. Such models have, to our knowledge, not yet been used in LCA of waste management.

3. Spatial information and information on specific pollutants

3.1. LCA characteristics and restrictions in applicability

Traditional LCA includes emissions and fuel demand of transports: it takes transport distances into account. However it does not differentiate between emissions occurring at different locations. Instead, all emissions of each specific pollutant are summarised, with complete loss of spatial information as a consequence. The environmental impacts of several pollutants may depend heavily on where and when they are emitted. As an example, the sensitivity for SO_2 emissions can be more than a thousand times higher in Sweden than in Greece (Hauschild and Potting, 2004), depending also on how the impact is defined. When geographical information is not included, the impacts of these emissions may not be accurately described. Because of its inability to handle spatial information, the typical LCA model also does not give information that is adequate for deciding where a waste-management facility should be sited.

Pollution involves a very large number of chemical substances. Society handles literally thousands of chemicals, many of them with largely unknown characteristics. Since these chemicals are used in different products, a very large number of chemicals will end up in the waste management system. The fate of these chemicals in different treatment processes is in practise impossible to model and include in an LCA. Furthermore, an LCA typically aggregates substances of the same type into sum parameters such as polyaromatic hydrocarbons (PAH), volatile organic compounds (VOC), and total organic compounds (TOC). Most probably, this is for practical reasons, since emissions are often reported in this manner in environmental monitoring. However, the environmental impacts may vary greatly between different substances within these sum parameters. Therefore, such aggregate measures reduce the ability of LCA to accurately model actual environmental impacts.

3.2. Amendments

The LCA can include a range of impact factors for each pollutant, corresponding to the spatial variability of the impact of the pollutant. With this approach, the LCA results will accurately reflect the uncertainty in actual environmental impact of a pollutant due to spatial variation; however, the approach will not reduce the potentially large uncertainty.

Approaches to reduce the uncertainty by taking geographical aspects into account have been presented for the assessment of several environmental impacts. It is useful to distinguish between site-dependent and site-specific modelling of the impacts (Hauschild and Potting, 2004). Site-dependent modelling takes into account the environmental conditions and sensitivity of the country or region where the pollutant is emitted. Site-dependent approaches have been developed for, e.g., acidification, terrestrial eutrophication, and tropospheric ozone formation (Potting et al., 1998a,b; Huijbregts, 1999; Krewitt et al., 2001; Hauschild and Potting, 2004). Site-dependent approaches are also integrated in recent LCA tools such as the EDIP 2003 (Hauschild and Potting, 2004). In the context of waste management, a site-dependent approach to acidification and human health was implemented by Finnveden and Nilsson (2005), to investigate whether a site-dependent approach would suggest a geographically differentiated national waste management strategy in Sweden. Nilsson et al. (2005) applied the site-dependent approach in an environmental assessment of a waste incineration tax in Sweden. They found that the level of impacts varied between different parts of the country, but this did not affect the ranking between the different waste management options.

Some site-dependent approaches tend to give a greater weight to pollutants that are emitted in regions where the level of pollution is already high. This encourages relocating activities that burden the environment to regions with lower levels of pollution. Hence, there is a risk that unreflective use of LCAs with a site-dependent approach results in the loss of relatively unpolluted areas.

Site-dependent approaches reduce the uncertainty of the environmental impacts caused by pollutants, but they do not include enough spatial detail to decide in what part of a region a waste-management plant should be located. The latter requires site-specific modelling, which takes into account the local conditions. Site-specific approaches have been developed for, e.g., the leaching of heavy metals from landfills (Hellweg et al., 2005) and for the impact of airborne emissions from, e.g., waste incineration on human health (Sonnemann, 2002). These approaches are so far rarely used in LCA case studies.

Alternative ways to obtain site-specific knowledge on the environmental impacts is by means of an environmental impact assessment (EIA) or risk assessment. These tools can take local aspects into account, and can be used for deciding what site for a waste management plant is best for the environment. An LCA can be included as part of an EIA, but the EIA also includes qualitative statements that can take into account for instance the specific value of unpolluted natural areas.

To increase the accuracy of the description of environmental impacts, some guidelines on LCA recommend that sum parameters should be avoided, and that data on emissions of specific substances should be used whenever possible. Guinée (2002) and other comprehensive guidelines also present characterisation factors for a great number of specific substances. A problem, in this context, is that emission measurements are often made using sum parameters. In these cases, data on emissions of specific substances do not exist. Because of the shear number of chemicals used in society we expect that there will always be data gaps for many chemicals that are potentially relevant in environmental assessments.

4. Non-linear relationships

4.1. LCA characteristics and restrictions in applicability

An LCA facilitates environmental comparisons of well-defined alternatives, such as recycling, landfilling and incineration of specific waste fractions. However, LCA models are typically linear steady-state models of physical flows (Guinée, 2002). The LCA results can indicate what waste-management option contributes the least to different environmental impacts. This is illustrated in Fig. 1a, which is a schematic representation of the weighted environmental burdens associated with the production and use of a hypothetical material. In this case, the results indicate that recycling is the environmentally preferable option because it reduces the total environmental impact.

In reality, the environmental burdens of collection and recycling are likely to be a non-linear function of the collection rate (see Fig. 1b). There will be initial activities and environmental burdens when a collection system is established. At very high recycling rates, the required extra transports and processing of materials may increase fuel consumption and emissions greatly for each additional tonne of material that is collected. The environmental optimal collection rate will be somewhere in between. However, since LCA results are linear, they cannot be used for identifying the optimum mix of waste-management options: recycling, landfilling and incineration. This means that typical LCA models cannot be used for identifying optimal reuse and recycling rates.

4.2. Amendments

Linear-programming (LP) models are linear models that account for boundary conditions. In waste management, limitations in achievable recycling rates of a bring system would be one such boundary condition. Very high recycling rates might require a switch to curbside collection, with higher economic costs and possibly more environmen-

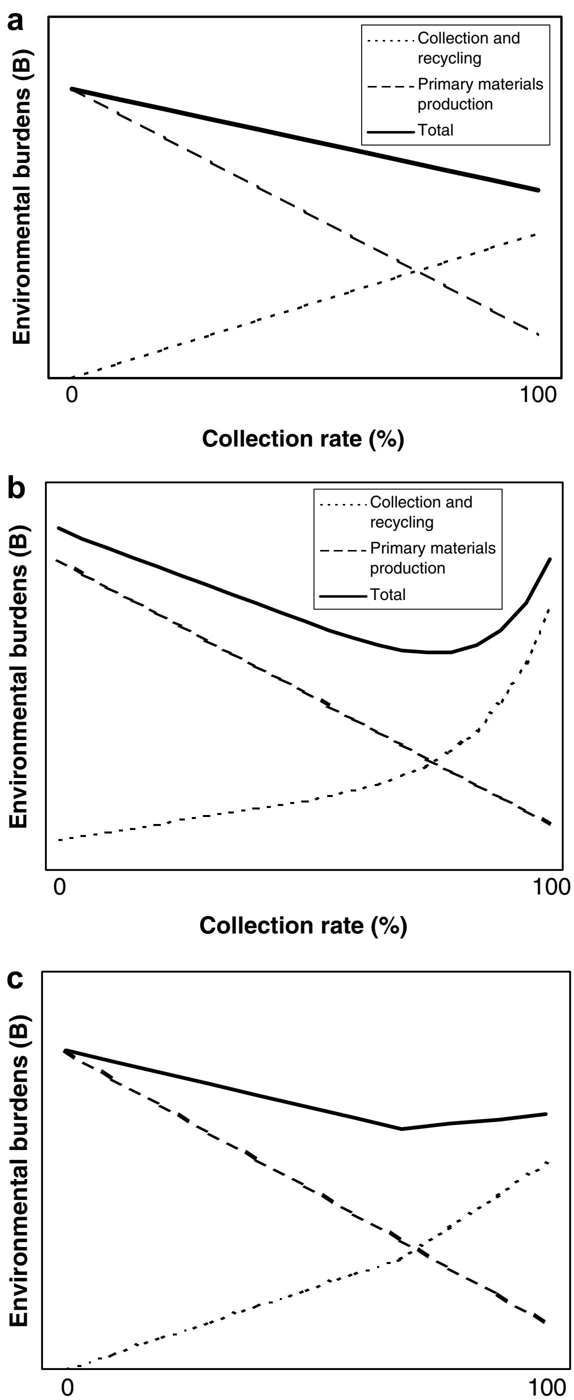

Fig. 1. (a) An LCA model typically describes environmental burdens of materials production as a linear function of the collection rate. (b) The environmental burdens of real collection and recycling schemes can be expected to be a non-linear function of the collection rate. (c) A linear-programming model can describe the system as a partially linear function of the collection rate.

tal burdens. As a result, the environmental burdens can be described as a partially linear function of the collection rate. As illustrated in Fig. 1c, such a function makes it possible to identify an optimal recycling rate. Optimising LP models of the waste management system can be integrated in an LCA. The ORWARE model and MIMES/waste are examples of LP model that integrate the life-cycle perspective and, hence, also are tools for LCA (Eriksson et al., 2003). A recent example that focuses on paper recycling was presented by Schenk et al. (2004).

Comparing Fig. 1b and c, it is easy to draw the conclusion that an LP model is not a very precise representation of the real system. Non-linear programming is required to account for the more complex, non-linear relations in the real system. However, as the complexity of the model increases, so does the requirement for data. High quality data for an LP model can be difficult to obtain. It is, for example, difficult to estimate the maximum collection rate that can be achieved through bring systems. The problem with data availability and data quality increases for a non-linear model.

5. Effects on background systems

5.1. LCA characteristics and restrictions in applicability

Many LCAs use average data to model the background systems, i.e., the systems indirectly affected by the actual system under study. In LCAs of waste management, important background systems include for instance the energy system and the production of materials and fertilisers, all of which may be significantly affected by decisions concerning waste management. The use of average data to model these systems may be relevant if the aim is to perform an attributional LCA (Tillman, 2000; Ekvall et al., 2004).

However, if the aim is to model the consequences of a decision, the use of average data may be misleading. The use of average data means that the LCA model is inaccurate in describing how the background systems are affected by changes in the waste management system, because changes in the waste management system will not affect all parts of a background system equally. For instance, changes in electricity use or generation in the waste management system will affect the electricity production system at the margin. In general, all actions in the waste management system can be expected to have marginal effects on the production of bulk materials (e.g., steel, aluminium, and polyethylene), energy carriers (e.g., electricity, fuel oil, and petrol), and/or fertilisers. Marginal effects are the consequences of infinitesimal or small changes in the quantity produced of a good or service (Ekvall and Weidema, 2004).

5.2. Amendments

Marginal effects should, ideally, be modelled using marginal data. These reflect, by definition, the environmental burdens of the technology affected by a marginal change (Weidema, 1993). If we account for the fact that a change in electricity use can affect investments in new power plants and the closing of old power plants, accurate identification of the marginal electricity production becomes difficult. The marginal electricity can be dominated by extended use of old coal-power plants, by the postponed closing of Swedish and German nuclear reactors, or by the construction of new CHP plants for natural gas, etc. Such effects are, in the context of LCA denoted as long-term marginal effects (Weidema et al., 1999).

The marginal technologies are often identified using static models of the electricity system, but they can also be analysed using dynamic optimising models. The latter approach gives a more complete description of the consequences of using or delivering electricity, because it takes into account effects on the utilisation of existing production facilities, as well as effects on investments in new production facilities. Mattsson et al. (2001) investigated how a dynamic optimising model of the production of electricity and district heat in the Nordic countries reacts to a change in the Nordic electricity demand or the Swedish nuclear power production. The results demonstrate that the marginal electricity production in the Nordic countries is complex in the sense that it involves several different technologies. The mix of technologies is uncertain because it depends heavily on assumptions regarding uncertain boundary conditions, future fuel prices etc. Scenarios from the study by Mattsson et al. (2001) were later used in an LCA on waste and competing fuels for Swedish production of district heat (Eriksson et al., 2007).

The LCA methodology can be further expanded to take more causal relationships into account and, hence, describe the consequences of a decision more accurately. Possible expansions include the integration of economic partial and general equilibrium models, experience curves, etc. (Ekvall and Weidema, 2004; Ekvall et al., 2004). A partial equilibrium model of scrap material markets has been presented by Ekvall (2000) and applied, for example, to model the consequences of cardboard recycling in an LCA of cheese (Berlin, 2002). This model takes into account the fact that the recycling of material from a specific product or a geographical area may affect not only the use of recycled material but also the collection for recycling of other products and in other geographical areas.

6. Non-environmental impacts

6.1. LCA characteristics and restrictions in applicability

The results of LCAs are limited to environmental impacts of waste management. Addressing the long-term sustainability of a waste management system requires knowledge of the financial costs and social impacts of available waste management options. Apparently, traditional LCA can only provide part of the necessary basis for a well-informed decision.

6.2. Amendments

It is possible to obtain a more comprehensive basis for decisions either by making separate analyses of financial costs (Thorneloe et al., 2007) and relevant social aspects or by expanding the LCA methodology to include these additional aspects. A study that includes financial costs as well as monetised environmental burdens, described through an LCA, is often called a cost-benefit analysis (Leach et al., 1997; Radetzki, 1999; Ekvall and Bäckman, 2001; Strömberg and Ringström, 2004). It has also been denoted as life cycle costing (Carlson Reich, 2005) or technology assessment (Assefa et al., 2005). These studies can also include other aspects such as the time required for source separation in households, the space required for the multiple dustbins used for the source separation, etc.

In these studies, the emissions and other environmental burdens are typically aggregated into one figure representing the environmental cost of each investigated option for waste management. This is made to be able to compare the environmental costs to the economic costs. The drawbacks are that a lot of information is lost in the aggregation and that the scientific basis for monetisation of environmental burdens has limitations (Stirling, 1997). The first problem can be partially overcome by not only presenting the aggregated results but also the disaggregated results from the life cycle inventory analysis and possible characterisation. The second problem can be partly amended by using several methods for monetisation in parallel.

7. Discussion

At first glance, the message of this paper may seem to be that LCA is quite insufficient as a decision-support tool in waste management. Our intention is, however, much more constructive. We believe that identifying its restricting characteristics, understanding the implications of these, and finding complementary tools, will lead to better use of LCA in waste management, either by actually finding ways of improving the models, or by simply being more realistic about their capacity.

Since different tools for environmental systems analysis are developed to focus on different aspects of reality (Wrisberg et al., 2002; Finnveden and Moberg, 2005), a combination of tools can provide a more holistic picture. Several of the limitations that are discussed in this paper are, however, general and relevant also for other tools for environmental systems analysis and, indeed, for science in general.

An LCA, just like systems analysis in general, entails a drastic simplification of the complex reality. The description can be more complete and detailed by adding methodological aspects: economic analysis, dynamic linear and non-linear modelling, site-dependent modelling of environmental impacts, etc. As more aspects are added to the analysis, the complexity of the study increases. More data are required, which increases the cost of the study. In order to provide the most comprehensive information possible about the consequences of possible actions – within the budget and/or time constraints given – the study should focus on the parts of the technological system that are expected to be most affected by such actions.

Several of the possible additions to LCA methodology require different types of economic data in addition to technological and environmental data. This means that economists ought to be involved in the study. Otherwise, the risk for mistakes increases. The data required to model the additional aspects are also often associated with a high degree of uncertainty. As a result, the uncertainty in the results of the study increases. It can be argued that, if an aspect of the reality is relevant to the study, it is better to describe it by using uncertain data than to ignore it completely. This implies that the boundaries of the study should ideally be defined at the point where the uncertainties and risk for mistakes become so large that further expansion of the study will yield no information that is significant for any realistic decision. Good judgement is required to identify this point in each case.

However, in our experience the audience or target group of a study tends to focus on the results of the study and disregard the significance of the uncertainties, even when they are clearly reported. If the study describes part of the reality with highly uncertain quantitative data, there is a risk that it will convey a false sense of security. This risk might be lower if highly uncertain parts of the reality are excluded from the study, provided that this limitation is clear from the report.

When the study grows increasingly complex, it also becomes more difficult to understand. This makes it more difficult for the target group to assess the credibility and relevance of the study. The transparency of the study can increase if different aspects are separately analysed - if the economic analysis, for example, is kept separate from the environmental assessment as far as possible. On the other hand, presenting them together makes it easier to find the most cost/eco-efficient solution. As a compromise between the difficulty of comprehending complex results and the need to "tell the whole story", information about uncertainties can be included on a demand-driven basis depending on the potential interest of the end users.

As evidenced by the suggested thematic strategy on waste by the European Commission (EC, 2005), there are great expectations on LCA and life-cycle thinking. Indeed, LCAs have been shown to provide policy relevant and consistent results (Finnveden and Ekvall, 1998; Björklund and Finnveden, 2005). However, it is also clear that the studies will always be open for criticism. Assumptions can be challenged and it may be difficult to generalise from case studies to policies (Finnveden, 2000). This suggests that there will continue to be a role for decision-makers in the policy process. If we have to wait for clear-cut and indisputable results from science, we may have to wait forever. If decisions are going to be made, they need to be made on a less than perfect basis. LCA and other tools for environmental systems analysis can contribute to the basis for such decisions, not by making it complete but by making it more comprehensive.

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