The MIIM LCA PhD Club

On the Limitations of Life Cycle Assessment and Environmental Systems Analysis Tools in General

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Abstract. The potential and limitations of life cycle assessment and environmental systems analysis tools in general are evaluated. More specifically this is done by exploring the limits of what can be shown by LCA and other tools. This is done from several perspectives. First, experiences from current LCAs and methodology discussions are used including a discussion on the type of impacts typically included, quality of inventory data, methodological choices in relation to time aspects, allocation, characterisation and weighting methods and uncertainties in describing the real world. Second, conclusions from the theory of science are practised. It is concluded that it can in general not be shown that one product is environmentally preferable to another one, even if this happens to be the case. This conclusion has important policy implications. If policy changes require that it must be shown that one product is more (or less) environmentally preferable before any action can be taken, then it is likely that no action is ever going to take place. If we want changes to be made, decisions must be taken on a less rigid basis. It is expected that in this decision making process, LCA can be a useful input. Since it is the only tool that can be used for product comparisons over the whole life cycle, it can not be replaced by any other tool and should be used. Increased harmonisation of LCA methodology may increase the acceptability of chosen methods and increase the usefulness of the tool.

Keywords. Allocation, case studies, characterisation, classification, data quality, decision support, environmental policy instruments, goal and scope definition, incineration, interpretation, landfilling, Life Cycle Impact Assessment, Life Cycle Inventory Analysis, methodology, system analysis, uncertainty, values, weighting

1 Introduction

1.1 Background

Life-cycle assessment (LCA) studies the environmental aspects and potential impacts of a product throughout its life from raw material acquisition through production, use and disposal (i.e. from cradle-to-grave) (ISO, 1997). "Products" are interpreted in a broad sense, and include both material products and services.

LCA is one tool used for describing environmental impacts. Examples of other environmental systems analysis tools include Risk Assessment, Environmental Impact Assessment (EIA), Environmental Auditing, Substance Flow Analysis, Energy Analysis, and Material Flow Analysis (see e.g. Anonymous, 1997; Wrisberg and Gameson, 1998; and Moberg et al., 1999 for a discussion on related tools. In this paper no distinction is made between procedural tools and analytical tools). What makes LCA unique is the "cradle-to-grave" approach combined with its focus on products, or rather the functions that products provide.

Although LCA is not something new, the interest has increased dramatically since approximately 1990, resulting in both a development and an increased harmonisation of methodology. A 'Code of Practise' has been published (Consoli et al., 1993) as well as several guidelines (e.g. Heijungs et al., 1992; Vigon et al., 1993; Lindfors et al., 1995a; and Wenzel et al., 1997) and an ISO standard (ISO, 1997). LCA is increasingly used by companies (Baumann, 1996; Berkhout and Howes, 1997; and Grotz and Scholl, 1996), and government agencies (Curran, 1997).

LCA has met high expectations but its results are at the same time often criticised (Udo de Haes, 1993; see also e.g. Ayres, 1995; Ehrenfeld, 1998; and Krozer and Vis, 1998). This criticism must be taken seriously in order to evaluate the role of LCA as a decision support tool for authorities and companies. LCAs may, for example, be criticised on the grounds that they do not produce the kind of information that is envisaged by the ambitious LCA definition. Important questions to be asked are therefore, what type of information can typical LCAs produce and for what purposes can this information be used. Another often encountered criticism of LCA is that it does not produce reproducible results. Comparative LCAs may sometimes lead to apparently conflicting conclusions. Due to the complexity of LCAs, it may be difficult to understand the reasons behind such differences. It is also often noted that the LCA methodology is still immature, under development and there is a lack of standardisation.

1.2 Aim of this paper

The aim of this paper is to evaluate the potential and limitations of LCA and environmental systems analysis tools in general. More specifically this is done by exploring the limits of what can be shown by LCA and other tools. This is done from three different perspectives. First, different aspects and limitations of LCA methodology are discussed. Second, experiences from case studies are used. Third, conclusions from the theory of science are practised. Some policy implications of the conclusions are also discussed. This paper is an elaboration of parts of my thesis (FINNVEDEN, 1998a), and largely based on my own research (FINNVEDEN, 1996a, 1997a and b; FINNVEDEN and EKVALL, 1998; FINNVEDEN and LINDFORS, 1998; FINNVEDEN and ÖSTLUND, 1997; FINNVEDEN et al., 1995). It should be noted that other examples and methodology aspects elaborated by other scientists could have been chosen.

2 On the Limitations of LCA and Related Tools

2.1 The question

Let us assume that there are two functionally similar products, A and B. Let us furthermore assume that there are some significant differences between them (which are larger than e.g. just a lower amount of a hazardous chemical in one of the products). The question that will be explored in this paper is whether it can be shown that one of the products is environmentally preferable to another one. It is argued that this will in general not be possible. In section 2.6 it will be argued that even if one product actually is environmentally preferable to another one, it will in general not be possible to show this. The wording "it can be shown" is interpreted as meaning that others can reproduce the results and conclusions and that other stakeholders will have to accept the conclusions.

Below, three different lines of argument are used, one is based on experiences from current LCAs and methodology discussions, the other two (the epistemological and the falsificationist's argument) are theoretical. The arguments are mainly focused on LCA. However, it is suggested that they will be equally valid also for other environmental systems analysis tools, hence the title of this paper. The arguments are presented below. First of all, however, let us consider what type of tool can be used in order to show that one product is environmentally preferable to another one.

2.2 LCA is the only available tool for products

If a proper comparison between two products is to be made, it will soon be realised that the whole life cycle must be considered. This is because different products may have burdens in different parts of the life cycle. For example, whereas one product may use less resources during the use phase, this may be at the cost of more resources used for production.

If the object under study is a product and the whole life cycle is considered, then it is a life-cycle assessment that by definition is performed. It is therefore difficult to imagine any other tool could be used in order to address the question concerning the environmental preference of two products. Of course, the tool would not have to be identical to current LCA methodology, but some sort of LCA would probably be necessary. Below, the possibilities for LCA to answer the question will be discussed.

2.3 Experiences from current LCAs and methodology discussions

In reviews and comparisons of different case studies (FINNVEDEN, 1997a; FINNVEDEN and EKVALL, 1998) it has been concluded that none of the discussed studies can be used to show the overall environmental preference for any of the alternatives compared. This is mainly because of three major types of reasons (which are further discussed below):

1) Not all relevant environmental impacts are considered.
2) There are uncertainties:
   a) in data,
   b) in methodology for the inventory analysis and the impact assessment, and
   c) in the description of the studied system.
3) The weighting element involves ideological and ethical values which can not be objectively determined.

2.3.1 Not all relevant environmental impacts are considered

One important choice when defining the scope of an LCA is the type of impacts included. According to the LCA definition, a study should include "the environmental aspects and potential impacts", suggesting implicitly that all relevant environmental aspects should be included. Within the ISO and SETAC frameworks it is also explicitly stated that the categories shall together enable an encompassing assessment of relevant impacts which are known today (UDO DE HAES et al., 1999). In operationalising this, different lists of impact categories have been suggested, e.g. the "check-list" from the Nordic Guidelines (LINDFORS et al., 1995a) and the default list suggested by the SETAC-Europe working group on life-cycle impact assessment (UDO DE HAES, 1996). These, and other, lists are similar, although not identical.

It is interesting to consider the impact categories which are normally treated in LCA. Based on the results of a review of studies comparing recycling with incineration of paper packaging materials (FINNVEDEN and EKVALL, 1998), the four case studies in the LCA-Nordic project (LINDFORS et al., 1995a) and a review of PVC databases (FINNVEDEN et al., 1996) the following general conclusions can be drawn.

- Energy as an input is included in most cases and without severe data gaps.
- Other raw materials are often covered but with severe data gaps. For example, raw materials to produce chemicals for the pulp processes in paper making are often missing (FINNVEDEN and EKVALL, 1998).
- Water is in most cases not included, and if it is, data gaps are often present.
- Land use, habitat alterations and impacts on biodiversity are in most cases not included, and if they are, data gaps are present.
- The human and ecotoxicological impact categories are often included but have severe data gaps. There is typically more data available for air emissions than for water emissions and the latter are sometimes completely lacking. The lack of toxicologically relevant parameters is also illustrated by the almost complete absence of information concerning the hundreds of additives and auxiliary chemicals used in pulp and paper production as well as in printing and de-inking.
- Non-toxicological human impacts and impacts in work environment are completely lacking in all reviewed case studies.

- Climate change is usually covered fairly well because CO_2 is usually fairly well covered. However, data for other pollutants, which in special cases can be of significance, is sometimes missing.
- Depletion of stratospheric ozone is often not relevant, but when it is relevant, data gaps are sometimes present e.g. in some PVC studies (FINNVEDEN et al., 1996).
- Acidification and eutrophication of terrestrial systems are often fairly well covered.
- Eutrophication of aquatic systems is often less well covered because of data gaps for water emissions.
- Photo-oxidant formation is normally covered to some extent but often has data gaps for organic compounds which nearly always is expressed as a general parameter.

The main conclusion from this section is that the LCAs studied do not cover all relevant environmental aspects. This limits the types of conclusions that can be drawn from these studies. No conclusions can be drawn concerning the overall preference from an environmental impact perspective of one choice over another, simply because all environmental aspects have not been included. The conclusions must be limited to aspects studied.

The situation described above is relevant for current LCAs. The situation in the future is likely to be ameliorated as better databases are developed. Although the situation will be improved, the problem of data gaps is likely to prevail for at least some of the impact categories. The human and ecotoxicological impact categories will probably never be described without significant data gaps. This is because a comprehensive evaluation is prohibited by the sheer number of chemicals used in society, combined with a lack of knowledge of the behaviour of all these chemicals in technical processes (see also TUKKER, 1998). It will for example, not be possible to analyse more than a fraction of the organic pollutants present in landfills (ÖMAN, 1998). Impact categories related to land use will continue to pose a problem for some time, partly because there is currently no agreement on how to describe land use in an inventory analysis. Current LCAs are thus focused on material flows. Other aspects, such as impacts, via land use, on the life-support systems providing ecosystem services, are largely neglected.

Before ending this section, it can be noted that the problem described here is not only a problem for LCA. Other types of environmental systems analysis tools also face similar, although perhaps not identical, problems in finding relevant data. For example, the discussion above on chemicals is valid also for other tools, e.g. Environmental Impact Assessments.

2.3.2 Quality of inventory data

The issue of data quality can be discussed based on a comparison between different databases for PVC (FINNVEDEN and LINDFORS, 1998, see also COPIUS PEEREBOOM et al., 1999). The databases are typical of the type of data a European LCA-practitioner would have chosen in the mid-1990s. The range of data in the different databases may therefore indicate the typical uncertainty. It is concluded that the uncertainties can be quite large, often an order of magnitude or larger (some rules-of-thumb are presented by Finnveden and Lindfors, 1998). In some cases, the large differences are likely to be due to mistakes (AYRES, 1995; FINNVEDEN and LINDFORS, 1998). Different allocation procedures can explain some of the differences but not in the order of magnitude in this example (BOUSTEAD, 1994). Differences in technology levels, existing at the same moment in the same country (or even at the same factory), can in some cases result in order of magnitude differences in emissions. One example for PVC production occurs with mercury emissions, where large differences can be expected if chlorine production with mercury cells is compared to other technologies. Examples for other materials can be found in Ekvall et al., 1992 and Ekvall, 1996. The data uncertainty can thus to a large extent be explained as real data variability due to different technologies. Instead of using the easily accessible databases, a careful selection of data for an appropriate technology level in relation to the goal of the study can therefore reduce data uncertainty (WEIDEMA et al., 1999). Another type of uncertainty is then introduced, concerning which technology is the appropriate one. This uncertainty is partly related to the description of the system discussed below in section 2.3.4.

The situation described by Finnveden and Lindfors (1998) concerns databases from the early and mid-1990s. Future databases will probably be improved. However, problems may still be created by methodological choices, differences in technologies used, as well as different types of knowledge gaps.

2.3.3 Methodology choices

Example 1: The time frame for landfills

One important difference between landfilling and most other processes in an LCA is the time frame. Emissions from landfills may prevail for a very long time, often thousands of years or longer. In order to make the potential emissions from landfilling comparable to other emissions during the life cycle, the potential emissions have to be integrated over a certain time-period. It is important to determine which time period is of interest. There is currently no international agreement on this question for landfills (FINNVEDEN and HUPPES, 1995). Using the LCA definition as a starting point, it can be argued that emissions should be integrated until infinity. In practise however, a shorter time frame (decades and centuries) has usually been chosen (see e.g. FINNVEDEN, 1999). Since both short and long-term perspectives seem to be of interest, two time perspectives were suggested by Finnveden (1992); the "surveyable time-perspective" corresponding to approximately one century and the "hypothetical infinite time perspective," defined as a complete degradation and spreading of all landfilled material. This approach has been further elaborated by Finnveden et al. (1995), Finnveden (1996a) Sundqvist et al. (1994 and 1997).

Does it matter which time frame is chosen? The answer is yes. The fraction of heavy metals in municipal solid waste, which is expected to be emitted during the surveyable time-period (approximately one century), is typically between 10^{-5} and 10^{-3} kg emitted per kg landfilled (FINNVEDEN, 1996a). In contrast, all heavy metals will be emitted during the hypothetical infinite time period by definition. Thus, the choice between a shorter time period (decades or centuries) and a longer time period can greatly influence the result.

The choice of the time frame is clearly a value choice for the inventory analysis. It is related to ethical views about impacts on future generations (FINNVEDEN, 1997b). A similar situation may occur for different parts of the impact assessment. The choice made by the SETAC-Europe working group on Life Cycle Impact Assessment is to consider first the infinite time period, then a short time period of 100 years and finally if wanted other time periods (UDO DE HAES et al., 1999).

Example 2: Multi-input allocation

Allocation problems in LCA have been much discussed (e.g. HUPPES and SCHNEIDER, 1994). One of them is the multi-input allocation problem. It occurs when several products are inputs to a process, and it focuses on determining which environmental interventions should be allocated to which products. An example is an incinerator for municipal solid waste which receives a large number of products and emits a number of pollutants, e.g. chlorinated dioxins to choose a much discussed example. An interesting question is then, how should the chlorinated dioxins be allocated among the incoming products?

There seems to be a general agreement in the LCA world that the guiding principle for the multi-input allocation should be the natural science based causalities (FINNVEDEN and HUPPES, 1995). This is also reflected in the ISO standard (ISO, 1998a). The question then becomes a scientific/technical question of finding the relevant causalities. To continue the example of chlorinated dioxins, it turns out that there are two positions that can be taken (FINNVEDEN and HUPPES, 1995):

Alternative A: Chlorinated dioxins are allocated to the incoming waste components in relation to their chlorine content.

Alternative B: Chlorinated dioxins are allocated to the incoming waste components in relation to their heat value, or something similar e.g. carbon content or flue gas volume.

There is support for both positions in the scientific literature. Several studies have investigated the correlation between the formation of chlorinated dioxins and the level of total chlorine in the fuel. Some studies have found a correlation and some have not (WIKSTRÖM et al., 1996). The choice between the two alternatives can have a significant influence on the results. It will for example influence the results of PVC, which contains chlorine. It will however have a much greater effect on other products, e.g. polyethylene (PE) and other plastics. If alternative A is chosen, no chlorinated dioxins will be allocated to PE, but if alternative B is chosen one kg of PE will cause more emissions of chlorinated dioxins than one kg of PVC.

This example shows that the allocation problem may still be difficult to solve, even though there is an agreement on the guiding principle. In the above example, one reason for the allocation problem is that the formation mechanisms of chlorinated dioxins are still not well understood (WIKSTRÖM et al., 1996).

The answer to the allocation problem may however also depend on the goal of the study. Different questions being asked may lead to different allocations. This is especially the case if the system behaviour is non-linear. The goals of LCAs can be analysed in different dimensions. A first fundamental dimension is concerned with whether the study is change-oriented or not (BAUMANN, 1998; FRISCHKNECHT, 1997 and 1998; HEIJUNGS et al., 1997a; TILLMAN, 1998; and WEIDEMA, 1998). With regard to the example given, one change-oriented question related to the consequences of a choice is: How would the emissions of chlorinated dioxins change if certain types of PVC were not incinerated? An example of a not change-oriented question is: Which fractions of the solid waste are responsible for the emissions of chlorinated dioxins? If the questions asked are change-oriented, other important dimensions concern the scale of the change and the time aspects. If the correlation between emissions of chlorinated dioxins and the level of chlorine in the fuel is non-linear, the scale of the change should influence the allocation. If the question is related to a small change (e.g. what is the effect of one extra kilogram of PVC in the incinerator?), the allocation should then be different than if the question is related to a large change (e.g. what would be the effect if PVC were banned?).

The uncertainties in this example are related to technical issues and the scientific understanding of the processes. It therefore illustrates the importance of knowledge gaps. LCA depends on other disciplines for data and methods. In the chlorinated dioxin example, the scientific community has difficulties providing a clear solution to the LCA problem. This is despite the large number of studies devoted to the problem of chlorinated dioxins produced when municipal solid waste is incinerated. Due to the large number of possible pollutants, there are probably many other substances for which the availability of data is much more limited. In the case of landfills, the scientific basis is perhaps even more limited. A major problem is that a long time perspective makes experiments and field studies difficult to perform. Another problem is the large amount of chemicals used in our society that ends up in landfills.

Example 3: Characterisation of abiotic deposits and other impact categories

There are several characterisation methods for abiotic deposits available (see e.g. FINNVEDEN, 1996b; and HEIJUNGS et al., 1997b, for recent reviews). They can give significantly different results (e.g. LINDFORS et al., 1995b; and FINNVEDEN, 1998b). Important questions are then: How can a choice be made between different methods and when is a specific method adequate? Unfortunately, there is no simple way of determining this (FINNVEDEN, 1994; and GUINÉE and HEIJUNGS, 1995). There is no method by which it can be shown that one characterisation method is the "correct" one. Instead, a theoretical reasoning must be applied. Underlying assumptions should be discussed. Examples of questions that can be raised are (FINNVEDEN, 1996b):

1) What is the problem with resource depletion?
2) Is the problem defined in a relevant way?
3) Does the quantification method adequately and reliably quantify the contribution to the problem?
4) Are there any logical contradictions?
5) Does the method produce reasonable results? Can we in any sense judge which results are reasonable?

We have earlier developed a new method for characterisation of abiotic deposits (FINNVEDEN and ÖSTLUND, 1997). It is based on exergy consumption, or entropy production, as a measure of the resource consumption. We suggest that this approach is based on a relevant definition of the problem (ibid.), and produces reasonable results. Others have reached different conclusions (e.g. HEIJUNGS et al., 1997b). This illustrates that the choice of characterisation method is a normative, value choice. Natural science alone can not provide the "correct" answer.

For other impact categories as well, the definition of the category indicator is a normative choice, although it should be based on knowledge of the relevant environmental problems. Within the ISO framework, all places in the environmental mechanism, are allowed for defining the category indicator (UDO DE HAES et al., 1999). In general, definition of the indicator closer to the environmental interventions (i.e. early in the cause-effect chain) will result in more certain modelling, but it may be less environmentally relevant. In contrast, definition of the indicator later in the environmental mechanism (i.e. closer to the category endpoints) will often make the modelling more uncertain but more relevant (FINNVEDEN et al., 1992; UDO DE HAES et al., 1999). The ideal place is thus a balance between different criteria. However, what is considered ideal by someone, may be considered less ideal by someone else. What is regarded as the ideal place is among other things influenced by the views on nature and the views on our ability to predict environmental impacts (FINNVEDEN, 1997b). People with a positive view of our abilities to predict environmental impacts may choose to define the category indicator later in the environmental mechanism, i.e. closer to the category endpoints. On the other hand, persons with a less positive view, perhaps emphasising the precautionary principle and the unknown parts of the environmental mechanism, will suggest that the effects are defined earlier in the environmental mechanism. This is because the unknown aspects will not be included if the indicator is defined later and if the unknown aspects are regarded as important it may be wise to define the indicator earlier.

It should be noted that the balance between known and unknown effects is not unique for LCA. This is probably an important aspect in all types of environmental assessment tools since the world is inherently uncertain.

General Comments

The examples above illustrate that there are methodological choices which have to be made and which introduces a certain uncertainty in the results. Some of these choices are value choices whereas others are more technical. There are value choices which has to be made both in the inventory analysis and in the impact assessment. These choices are influenced by cultures, frames and paradigms (HOFSTETTER, 1998; TUKKER, 1998). Different methodological choices will be more or less compatible with different frames and cultures, leading to different choices of methods and tools by different people (ibid.).

One way of reducing the uncertainty due to choices is through harmonisation and standardisation (HUIJBREGTS, 1998). Such efforts have already taken place in several international fora resulting in substantial advancements in the harmonisation and standardisation of LCA methodology (e.g. CONSOLI et al., 1993; LINDFORS et al., 1995a; UDO DE HAES, 1996; ISO, 1997, 1998a, 1998b; UDO DE HAES et al., 1999) and future work in this direction should be supported. Since agreements can be challenged by scientists, the question is whether a sufficiently practical and scientific harmonisation can be developed.

2.3.4 Uncertainties in describing the real world

In a review of studies comparing recycling to incineration of paper packaging materials, it is concluded that the results largely depends on some key aspects of the surrounding systems (FINNVEDEN and EKVALL, 1998). One key issue is the fuel being incinerated for heat production, if the paper is recycled. Other important aspects concern the electricity production, the heat sources at the mills and what happens to the wood which is "saved" if the paper is recycled (ibid.). In a descriptive, not change-oriented study, these aspects may in principle be possible to find out, although in practise it may prove to be difficult. However, in a change-oriented study, aspects like these are inherently uncertain. This is because the future is inherently uncertain. The uncertainty may be reduced with different kinds of future oriented studies, but can never be eliminated.

2.3.5 Weighting methods

The weighting can be defined as the qualitative or quantitative element in which the relative importance of different environmental impacts is weighted against each other (UDO DE HAES, 1996). The weighting can be performed on a case-by-case basis by an LCA practitioner or a panel or by using generic weighting factors.

It is generally recognised that the weighting element in LCA requires political, ideological and/or ethical values and these are influenced by perceptions and world views. Examples of questions that can be raised are (FINNVEDEN, 1997b):

- Are future people moral objects and if so, how important are they?
- Are animals, plants, and/or ecosystems also moral objects?
- To what extent are we able to predict environmental impacts?
- What is the importance of natural systems in relation to economic systems?

Answers to these questions, and others, will have an influence on the weighting. Not only the weighting factors, but also the choice of weighting methodology, and the choice of using a weighting method at all, are influenced by fundamental ethical and ideological valuations (FINNVEDEN, 1997b). Since there is no societal consensus on these fundamental values, there is no reason to expect consensus either on weighting factors, or on the weighting method or even on the choice of using a weighting method at all. As noted above (and also discussed by HOFSTETTER, 1998; and TUKKER, 1998) some fundamental values can also have repercussions on methodological choices for the inventory analysis, and the classification and characterisation elements of the impact assessment.

It can be noted that there are cases in which all regarded aspects point in the same direction and for which no weighting is therefore necessary. In these cases, this argument will not be valid. However, these cases will probably remain exceptions. Since there is no way by which we can evaluate the "true" values, results which rely on weighting can always be challenged by other stakeholders. It is however important to note that if it is not necessary that other stakeholders accept the conclusions, a decision-maker can base the conclusions on a specific set of values, and in these cases, this argument is not relevant. Also if the involved stakeholders can agree on a specific set of values beforehand, the argument can be circumvented.

2.3.6 Future development

The arguments above concern limitations of current LCAs. It is clear that the studies can be improved in various ways. It is however difficult to see how the limitations could be completely avoided. For example, although data gaps and data uncertainties are likely to be reduced, they will still remain. This is especially the case for pollutants with potential human and ecotoxicological effects. This is because the shear number of chemicals used in the society today will prohibit comprehensive assessments. Another inherent limitation concerns the modelling of the system which especially for change oriented studies will remain inherently uncertain. This is because the future is inherently uncertain.

Once again it can be noted that many of the limitations are not specific to LCAs but are generic to environmental systems analysis tools in general.

2.4 Theoretical arguments

2.4.1 The epistemological argument

The epistemological argument is taken from Heijungs (1997 and 1998). We can not empirically study the environmental impacts of a single product throughout its life cycle. Since impacts that are observed in the world can not be connected to products by an experimental method, one must rely on models, that are only valid within a certain context (HEIJUNGS, 1998). The models will be based on some postulated properties, definitions and axioms, which themselves can not be proven. Equally valid models, giving equally valid results, can therefore be developed from different starting points. It can therefore not be shown which method, or which result is the correct one.

The importance of this argument can of course be reduced if agreements are made on the postulated properties from which a common methodology could be developed. Bearing in mind the possible goals of LCA (e.g. change-oriented studies or not), several methodological frameworks can be developed (HEIJUNGS, 1997).

The epistemological argument was developed by Heijungs (1997) for descriptive LCAs, and not for change-oriented LCAs. However, we can not perform experiments at world level including all process and involving time integration. It is therefore not possible to perform experiments to verify the change-oriented LCAs either and they too will rely on models, although technical elements of the models can be validated. It is therefore suggested that the epistemological argument is valid also for the change-oriented LCAs.

2.4.2 The falsificationist's argument

The result from an LCA is a single observation statement. It is single because the model was run with a certain set of data and certain system boundaries. We can rerun the model using another set of data or slightly different system boundaries, resulting in a new observation statement. However, a statement that one product is environmentally preferable to another one is a universal statement. The question is therefore, how can we come to a universal statement from single observations? From the theory of science (e.g. CHALMERS, 1982) we can learn that universal statements are logically impossible to prove. We can only put them up as hypothesis or theories. These can be falsified but never proven to be correct. Anybody who wants to challenge the results from an LCA can always ask for a new situation with slightly different properties that were not included in the original calculations. The prudent LCA practitioner can in this situation be forced to answer: No, we did not include that specific situation and we can therefore not draw any universal conclusions. The importance of the argument can of course be reduced with improved studies and an increased number of scenarios studied, but it can probably not be eliminated. It should also be noted that for specific conditions, specific conclusions may be drawn which of course can be useful.

2.5 The retreat to possibilities

The possibility of a retreat to probabilities should also be examined (CHALMERS, 1982). Perhaps the statement could be refined to "One product is environmentally preferable to another one with a probability of X %". The use of statistical methods can certainly be useful. However, since some of the systems uncertainties are large and involve aspects such as reliability of theories and information as well as ethical and epistemological aspects, uncertainty can not be handled at the technical, routine level (FUNTOWICS and RAVETZ, 1994). There is no way the uncertainty in for example the choice of methods for the impact assessment can be described in a statistical distribution. Also, uncertainties in the described system (e.g. what will the waste management system look like in 15 years) are difficult to describe in statistical terms. Also data gaps can not be described by statistical uncertainties. Data uncertainties can and should be described with statistical methods, but the other types of limitations discussed will be more difficult. The switch to a statistical formulation will therefore only partly solve the problem.

2.6 A thought-experiment and conclusion

Above it was argued that it will in general not be possible to show that one product is environmentally preferable to another one. This does, however, not eliminate the possibility that one product actually is environmentally preferable to another one. Let us for the sake of the argument assume that product A actually is environmentally preferable to product B in the sense that it would be environmentally preferable if we choose product A instead of product B. Would this then be possible to show? First it can be noted that the assumption that product A actually is environmentally preferable to product B implicitly assumes either that there are some methodological choices and value choices which are the correct ones or that they are of limited importance. Let us consider the first case first. Even if there are some choices which are the correct ones, it is not possible for us to show which of these are the correct ones, and we still have to face the same uncertainty. All the arguments raised above are therefore still valid. Thus, even if one product actually is environmentally preferable to another one, this will not in general be possible to show. If the choices are of limited importance, this will mean that some of the arguments raised above will be of limited importance. The other arguments will however still remain.

The conclusion from this section is that it will in general not be possible to show that a product is environmentally preferable to another one, even if this happens to be the case. This is not possible with LCA or any other related tool.

3 Some Implications

The conclusions drawn in the preceding section have implications for policy making. If regulations or international agreements require that it must be shown that one product is more (or less) environmentally preferable to another before any action can be taken, then it is very likely that no action will ever take place. This is because it will not be possible to show this (as concluded above), even if it happens to be the case that one product is environmentally preferable to another. If the society wants to be able to change policy or take action, e.g. by labelling or regulating the use of a specific type of product, then it must be possible to take decisions, even if the scientific basis for the decisions is not as rigid as one could wish for.

Another implication is that LCAs are useful as a defensive and conservative tool. If a product, or material, or policy is under attack, it will often be possible to perform an LCA, or a similar type of study, which can confuse the debate. The company or authority under attack can claim that no action should be taken unless the alternatives are shown to be better. Since this is normally not possible (according to the conclusion above), this can be an effective strategy. Industries have often used LCA defensively (Berkhout and Howes, 1997) and this is perhaps one reason why environmental NGOs have often been sceptical about LCA (Elkington, 1993).

LCA can be used for both qualitative and quantitative comparisons. The conclusions drawn in the interpretation phase of an LCA will depend on the questions being asked when the goal is defined. The example of flooring materials discussed by Finnveden (1997a) can be used to illustrate this. If the question asked is: Can it be shown that any of the alternatives to PVC flooring are environmentally preferable, then the answer according to the discussion above, is probably no, even if it would be the case, and a possible conclusion can be that there is no reason from an environmental perspective to change flooring material. If on the other hand, the question asked would be something similar to, "PVC flooring contains some additives we would like to avoid and there are also some problems in the waste management phase of the life-cycle, but before deciding on switching to another material we would also like to consider the whole life-cycle of PVC and its alternatives in order to check that we do not run into any new problems". The answer from such a question would then be something like: "Although there are uncertainties and data gaps, there does not seem to be any major new problems associated with polyolefin floorings compared to PVC and they could therefore be chosen instead. In the case of linoleum floorings however, there are issues concerning the pesticides used and also for the waste management phase which should be further checked before a switch is made." Apparently very different conclusions can thus be drawn from the same study depending on the questions being asked.

According to the conclusions above, questions of the type: Which product is environmentally preferable, A or B, are in general impossible to answer, regardless of method. If the study is to be used for anything but conservative purposes, care should be taken when defining the goal so as to formulate questions and hypothesis that can be answered. Since the questions and hypothesis often are formulated together by the LCA practitioner and commissioner, they have a joint responsibility. The opportunity for others to reuse the study, asking new questions, should however not be overlooked.

It should be noted that the implications of the conclusions above can sometimes be very small. LCA can be very relaxed (Udo de Haes, 1998). This is especially the case if the LCA is used mainly for learning, or internally within an organisation. The starting point for the discussion was a question in section 2.1 in which the wording "to show" was given a rather strong interpretation assuming that other stakeholders would have to accept the conclusions. If this requirement is loosened, the conclusions will change. For example, if "to show" means that standard methodology has been used which is accepted within a certain group of people and organisations, then many of the arguments raised above will not be valid and the usefulness of LCA may increase. This is one reason why it is important to strive for an increased harmonisation of the LCA methodology, also for the weighting methods.

The limitations discussed above can however become evident if the tool is used to support decisions of commercial or political importance and when there are organisations involved which are strong enough to challenge the basis for the decisions.

4 LCA as an Environmental Systems Analysis Tool

The limitations discussed in section 2 were discussed with LCA as a starting point. However, most of the discussion is valid also for other types of environmental systems analysis tools. Questions concerning data gaps, data uncertainties, methodological choices, values, and description of the studied systems are relevant also for other tools, as well as the theoretical arguments.

Environmental systems analysis tools have of course similar potentials and limitations as systems analysis tools in general. It can therefore be interesting to compare LCA to other systems analysis tools. The purpose of systems analysis is to help a decision maker to choose a better course of action in a particular problem situation than the decision maker might otherwise be able to act (QUADE and MISER, 1985). To be useful, however, the analysis does not have to provide a complete prescription as to what should be done (ibid.). In truth it cannot; the uncertainties are usually such that, while the analyst may aim to produce facts and proofs, the results are merely evidence and arguments (ibid.). I believe that the arguments provided in section 2 suggest that LCA is not significantly different from other types of systems analysis.

5 On the Usefulness of LCA

Is LCA of any use if the aim is to reduce the overall environmental impacts of society? I think the answer is: Yes, it can be. If society is to reduce its overall environmental impacts, it is necessary to discuss and improve the environmental performance of the products used to fulfil different functions. In order to do this, tools to describe the environmental impacts of the products are needed. Sometimes it will be useful to address the whole life cycle of the products, in order to avoid a situation in which problems are shifted between different life-cycle phases. LCAs will therefore be useful. Other tools should of course also be used. LCA does, for example, not normally address the number of products (or functions) used in society, which is also of importance. But LCA is unique in its focus on products and the "cradle-to-grave" approach, so LCA can not be completely replaced by any other tool.

As illustrated in reviews of case studies, LCAs can have policy implications and help different types of decision makers, e.g. involved in purchasing (FINNVEDEN and EKVALL, 1998; FINNVEDEN, 1997a). It has also been exemplified how LCA can be used to identify areas for improvements of product systems. The study on paper recycling (FINNVEDEN and EKVALL, 1998) was used as one of several background documents in Swedish policy making on reuse of packaging materials (NATURVÅRDSVERKET, 1998). Among the results of the study that were used in the discussions and conclusions were the total energy use, the use of fossil fuels, emissions of greenhouse gases, other air pollutants and amount of landfilled solid waste. The LCAs could also be used for improved policy discussions, and ultimately decisions, by helping to identify aspects of the systems which are important for the outcome (in this case the fuel competing with the paper), and equally important by identifying aspects which are of limited importance for the results (in this case collection and transportation of waste paper as long as it is reasonably efficient).

The letter on flooring materials (FINNVEDEN, 1997a) was written as a comment to an original study by Günther and Langowski (1997). One of the useful results of their study was the identification of important areas for improvements. Examples included changes in the composition of flooring materials, increased use of recycling materials, and recovery and reuse of wasted floorings (GÜNTHER and LANGOWSKI, 1997). Although not intended for that purpose, the results can also be used for policy making and by decision-makers involved in purchasing. For example, policies supporting the increased use of recycling materials, and recovery and reuse of waste floorings can lead to improvements. Also, the results discussed by Finnveden (1997a) can together with a substitution principle be used to argue that PVC floorings should be avoided and polyolefin floorings used instead.

It has been suggested that LCA probably has its best use as a tool for learning rather than a tool for supporting specific decisions (BAUMANN, 1998). This is in line with the discussion above where it was concluded that LCAs normally can not produce conclusions of the type "Product A is environmentally preferable to product B" even if this happens to be the case. However, it is expected that LCAs can increase the environmentally related knowledge of the studied systems, identify critical parts (or "key issues"), and separate important parts from less important parts. Decisions will be taken anyway, and it is expected that LCAs in many cases can provide a better basis for the decision making process. In fact it is the only tool available for making comparisons of products considering the whole life cycle, and it should therefore be used for that purpose. It is believed that it can provide crucial information for the decision making process, leading to a better course of action than the decision maker might otherwise have chosen. In order to make full use of the tool, both quantitative and qualitative comparisons should be made.

The mere use of LCA will however not lead to reduced environmental impacts. As discussed above, LCA can be used as a conservative tool to obstruct changes that would be environmentally beneficial. Since it in general will not give answers of the type "Product A is environmentally preferable to product B", it is important not to be paralysed with the lack of clear prescription. Some of the properties of LCA, e.g. the possibilities to identify data gaps and uncertainties, will often work conservatively. When formulating the hypothesis and defining the goals of a specific case study, and when using the results, there will be a possibility to use the results progressively or conservatively. In each case study, LCA practitioners and commissioners have the responsibility of making this choice.

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News and Views: Environmental Life Cycle Assessment of Linoleum

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Introduction and goal

Forbo-Krommenie B.V. commissioned the Centre of Environmental Science (CML) to carry out an Environmental Life Cycle Assessment. The purpose of this study was to assess the environmental performance of linoleum floors, indicating possible options for improvement, and assessing the sensitivity of the results to methodical choices. The method followed in this study is based on Guinée et al. (2000) an update of the CML guide on LCA from Heijungs et al. (1992).

The functional unit was defined as: 2000 m^2 linoleum (produced by Forbo-Krommenie B.V. in 1998) used in an office or public building over a period of 20 years.

Results and conclusions

The growing of linseed turned out to be the process contributing most to many impact categories. Other impoertant proccesses were:

- Oil and coal used for the production of maintenance products.
- The transport of raw materials.
- The incineration of linoleum.

Scenario analysis showed that uncertain data such as the pigments used and the type of VOC emitted can have a substantial influence on the outcome.

The major data gaps in the study are capital goods and chemicals (chemicals used for maintenance products, pesticides, catalists, etc.). Sensitivity anaysis showed that these gaps can lead to an underestimation of 1-10% for missing capital goods and 5-40% for missing chemicals. Therefore, the results should not be used to compare different production systems.

Based on the study, some options to improve the environmental performance of Forbo-Krommenie B.V. were formulated and also advise for further studies on linoleum was given.

Copies can be ordered as follows: T (+31) 71 527 74 85; F (+31) 71 527 55 87; e mail: eroos@rulcml.leidenuniv.nl; CML Library, P.O. Box 9518, 2300 RA Leiden, The Netherlands. Please mention report number 151 , and name and address to whom the report is to be sent.