Quantitative life cycle assessment of products

2. Classification, valuation and improvement analysis

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In a previous article about life cycle assessment (LCA), a methodological framework was proposed and two components of this framework were discussed in more detail: the goal definition and the inventory. In this second article, the other components of the framework are discussed in detail: the classification, the valuation and the improvement analysis. In the classification, resource extractions and emissions associated with the life cycle of a product are translated into contributions to a number of environmental problem types, such as resource depletion, global warming, ozone depletion, acidification, etc. For this, each extraction and emission is multiplied with a so-called classification factor and the multiplication results are aggregated per problem type. Classification factors are proposed for a number of environmental problem types. The valuation includes both a valuation of the different environmental problem types and an assessment of the reliability and validity of the results. For the valuation of the environmental problem types, qualitative or quantitative multicriterion analysis could be applied. Given a standard list of weighting factors the quantitative multicriterion analysis seems preferable, because of its low costs and its simplicity. The main problem, however, is to get a broadly supported standard list. In studies so far little attention is paid to the assessment of the reliability and the validity of the results. To improve this situation methods which could support this assessment are proposed. In the improvement analysis potential options to improve the product(s) studied are identified. Combined with expertise in other fields, such as costs and technological feasibility, the improvement analysis may yield a number of serious options for the redesign of a product. Two complementary techniques for the identification of the potential options are discussed. With these techniques and the active participation of process technologists and designers, LCA might become an analytic tool for eco-design supporting a continuous environmental improvement of products.

Keywords: life cycle assessment; environmental management; classification

Introduction

This is the second article dealing with quantitative environmental life cycle assessment (LCA) of products. In the first article[1] it was argued that LCA can become an important tool in product-oriented environmental management. It was concluded that current methods are divergent, yield conflicting results and contain considerable gaps. To enable fruitful discussions on methods used, and to make LCA an acceptable tool for environmental management, a general methodological framework was proposed, consisting of five components: goal definition; inventory; classification; valuation; and improvement analysis†. The first two components of this framework were discussed in detail in the first article. A short summary of the content of these two components is given below.

In the goal definition the goal of the study is defined in relation to the application intended. The type of application will influence the whole procedure. The applications always involve some kind of comparison both with product (system) comparison and with product (system) improvement. Then a unit of use should be specified forming the basis for comparison. This unit is based on the function of the products to be compared, and is called the functional unit of a product. Also the spatial scale and the time horizon of the study are determined in the goal definition.

In the inventory, the life cycle is the guiding

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† The framework proposed and the terminology used in this article differ in some cases from those proposed at a recent workshop in Lisbon organized by SETAC (Society of Environmental Toxicology and Chemistry). During this workshop a draft 'Code of Practice' for LCA-practitioners has been prepared. In this Code of Practice the following technical framework is presented: 'Goal Definition and Scoping' (here Goal Definition), 'Inventory Analysis' (here Inventory), 'Impact Assessment' consisting of the following steps: 'Classification', 'Characterization' (here together Classification) and 'Valuation' (here Valuation as a separate component), and 'Improvement Assessment' (here Improvement Analysis). Due to the date of submission of this article, the Lisbon framework and terminology could not be followed yet.

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principle. The product during its entire life cycle from cradle to grave, in terms of all related economic processes, is called the product system. The term economic process refers to any kind of process producing an economically valuable material, component or product or delivering an economically valuable service including transport and waste management. To make a quantified survey of the environmental inputs and outputs of a product system, the boundaries between the product system and the environment and the boundaries between the product system and other product systems must be determined. The latter is as yet undefined in three cases: production of co-products, combined waste processing, and open-loop recycling. For these three cases several methods are possible for the allocation of the environmental impacts, which have been discussed in the previous article. Furthermore, it also has to be determined in the inventory how a cut-off can be made between relevant and less relevant processes related to the product system. Last but not least, process data have to be gathered. It was concluded that the limited availability of data is still one of the major problems related to the inventory. The result of the inventory can be called the inventory table, a list of inputs from (resource extractions) and outputs (all kinds of emissions) to the environment.

The general question addressed in this article is how to proceed after the inventory. This topic has received a lot of attention lately[2–9], but it has also caused extensive discussions. Thus, maybe another question should be asked first: why should in LCA any further components be included at all? The major reason is that emissions (of substances) and extractions (of resources) as listed in the inventory, have no meaning in themselves. It is the problems caused by these extractions and emissions which are important. Basically it is true that, if in a comparison the data on all emissions and extractions point in the same direction, further analysis of these problems is not necessary to reach a decision. In most cases, however, one product is better on some extractions and/or emissions and worse on others; and the same may hold true for product improvement. Then further information about the relation between the extractions and emissions and the environmental problems is needed.

In this additional information both knowledge about environmental processes and effects and social weighting processes play a role. Two approaches are possible. The first is that these two elements are combined in one single methodological component as is proposed by Ahbe *et al.*[2], by Ryding[4] and by Krozer[10]. In these methods all environmental inputs and outputs of the inventory table are aggregated in one step into one overall score. The second approach is that the two elements are separated and dealt with in two successive components, i.e. the classification and the valuation. In the USA, classification and valuation are described under the heading impact analysis[11]. One of the main arguments for this separation is that each element needs its own expertise. Thus, in the classification extractions and emissions are aggregated per type of environmental problem, applying as much scientific knowledge as possible about environmental processes and effects. Or in other words, in the classification extractions and emissions are aggregated on the basis of their potential effect on a number of assessment endpoints (problem types). In the valuation different problem types are weighted against each other based on social values and preferences. As already presented in the previous article, we propose to distinguish these two components.

Classification

The aim of the classification is now defined as: to quantify the contribution of environmental inputs and outputs of a product system: to a number of generally recognized environmental problems; per problem type; and taking into account all relevant environmental processes. The result will be an aggregation of the large amount of data of the inventory table into a number of so-called effect scores.

As far as potential health problems of emissions are concerned, present practice is based on a media-oriented approach and normative environmental standards such as MAC-values (maximum accepted concentrations; these are on-site industry standards). This results in critical volumes of air and critical volumes of water[12,13] or units of polluted air and units of polluted water[14,15], which amounts to the same. The procedures followed in these approaches are essentially the same; the emissions are added up after first having been divided by quality standards for human health. Only the standards used do differ. The Dutch studies[14,15] use MAC-values for the aggregation of airborne emissions, and EC-directives for surface water intended for drinking water production for the aggregation of waterborne emissions. The Swiss studies[12,13] use German MIC-values (maximum immission concentration; maximale Immisionswerte des Vereins Deutscher Ingenieure) for the aggregation of airborne emissions, and Swiss directives for emissions into surface water for the aggregation of waterborne emissions. Other studies add emissions by mass without any further assessment[16].

The definition of the classification as proposed above differs in two ways from this current practice. On the one hand, a problem-oriented (cross-media) approach is proposed in contrast to the current media-oriented approach. The problem-oriented approach is preferred because it gives better possibilities for a scientifically based classification due to the greater similarity between the environmental processes involved, and it has a more direct relation with present day environmental policy, which is also increasingly problem-oriented. On the other hand, classification and valuation as defined above make a further distinction between environmental and social aspects, thus distinguishing between two different fields of knowledge.

Methods for classification

In February 1992 at a workshop in Florida, a general discussion was held on methods for classification of emissions of substances[11]. From the discussions between participants, five possible methods for classification of emissions of substances were proposed:

1. loading assessment, aggregating both waterborne emissions and airborne emissions separately to their mass (kg) without any further assessment

2. impact equivalency assessment, aggregating emissions to their potential effects without any exposure analysis
3. toxicity, persistence and bioaccumulation profile approach, aggregating emissions separately to their inherent toxicity (=potential effect), persistency and bioaccumulating behaviour
4. generic exposure-effect assessment, aggregating emissions based on a generic (not site specific) analysis of the exposure and effects due to a particular emission, sometimes taking into account generic background concentrations
5. site-specific exposure-effect assessment, aggregating emissions based on a site specific analysis of the exposure and effects due to a particular emission taking into account site specific background concentrations.

This seems to give a large choice in classification methods. However, looking in more detail at these five methods, the choice proves to be smaller. In particular, the loading assessment does not meet any of the elements of the classification definition. In this method the principle 'less is better' is applied without an assessment of the different environmental effects of the inputs and outputs. In fact, it concerns a grouping of the data of the inventory table without further analysis. A site-specific exposure-effect assessment on the other hand is not practicable in an LCA which generally is about dozens of processes all over the world. This method is more appropriate in an environmental impact assessment (EIA), where an environmental analysis generally is performed for one activity at a well-defined site. Consequently, only the impact equivalency assessment, the toxicity, persistence and bioaccumulation profile approach and the generic exposure-effect assessment are left.

An impact equivalency assessment only deals with potential effects on endpoints without regarding preceding environmental processes. Examples of this approach are the critical volumes approach as mentioned above, and the approaches for acidifying and nutrifying emissions (see below). Example of a toxicity, persistence and bioaccumulation profile approach is the Swedish proposal for the assessment of ecotoxic substances[8] which includes an assessment of the inherent toxicity by means of an LC_{50} an assessment of the bioaccumulation by means of a so-called bioconcentration factor and a qualitative assessment of the persistency of a substance by classifying them into 'readily' or 'not readily biodegradable'. This results in four partial effect scores for ecotoxic substances[8]. Examples of the generic exposure-effect assessment are the classification of ozone-depleting emissions according to ozone depletion potentials (ODP), the classification of greenhouse emissions according to so-called global warming potentials (GWP), and the classification of photochemical oxidants creating emissions according to so-called photochemical ozone creation potentials (POCP). These methods result in one general effect score per problem type.

The impact equivalency assessment, the toxicity, persistence and bioaccumulation profile approach and the generic exposure-effect assessment are three possible methods for classification of emissions of substances. We think that the generic exposure-effect assessment resulting in one effect score per problem type is the preferred method for LCA, while the impact equivalency assessment and the toxicity, persistence and bioaccumulation profile approach can offer a temporary solution as long as a generic exposure-effect assessment is not yet feasible. The problem of the toxicity, persistence and bioaccumulation profile approach is of course how to weight the different aspects against each other.

In the classification four steps can be distinguished:

1. the definition of environmental problem types;
2. the definition of classification factors indicating the contribution of one unit of an environmental input or output to each of the environmental problems to be defined;
3. the multiplication of environmental inputs and outputs with their classification factors and aggregation of the results per problem type into a number of effect scores; and
4. the normalization of effect scores.

Some of these steps (1 and 2) are of a more methodological nature and others (3 and 4) are more practice-oriented. Below, we will offer proposals for the elaboration of the methodological aspects of these steps. In principle, a performer of a case study would only have to consider the practical aspects. However, we realize that the methodological proposals to be discussed here will not be suitable for every conceivable case study and will need further improvement. Therefore, there should be a clear opportunity for an LCA-performer to adapt the methodology. The methodological aspects of each of these steps will be discussed subsequently.

Environmental problem types

First, a list of generally recognized environmental problems in terms of assessment endpoints for the classification, should be defined. Environmental problems can be expressed, as also suggested by Finnveden *et al*[8], at different levels of the environmental effect chain. As an example, *Figure 1* shows the effect chain for global warming.

Global warming is caused by emissions of different

Emissions of greenhouse gases

↓

Change in radiative forcing (1st order effect)

↓

Global temperature change (2nd order effect)

↓

Rise of sea level due to sea water expansion, melting of ice caps (3rd order effect)

↓

Rise of sea level due to ice melting (4th order effect)

↓

Degradation of ecosystems (5th order effect)

↓

etc

Figure 1 Environmental effect chain for global warming[8] substances, the magnitude of which are determined in the inventory. These substances all absorb infra-red radiation, which results in a disturbed balance between the energy absorbed by the earth and the energy reflected. This change in radiative forcing of the atmosphere is called the 'greenhouse effect' and can be characterized as the primary effect in the effect chain. It is assumed that this change in radiative forcing will change the global temperature (secondary effect), which in turn can result in a rise of the sea level due to sea water expansion and a melting of the ice caps (tertiary effect), a rise of the sea level due to ice melting (quartiary effect), degradation of ecosystems (quintary effect) etc.[5,8].

Moreover, all kinds of feedbacks are possible within one effect chain or between different effect chains. For example, emissions of volatile organic compounds (VOCs) can form ozone in the troposphere under particular meteorological circumstances. Ozone, for its part, can contribute to global warming. Thus, ozone formation due to an emission of a particular VOC is an effect in one effect chain and at the same time an input for another effect chain.

The possibilities to predict effects decrease as the order of effects increases. In principle, inputs and outputs should be linked to the lowest order effect, which can still be clearly related to the effect chain considered. Thus, global warming in terms of a change in radiative forcing should preferably be chosen as the assessment endpoint of the classification, although it is only an intermediate point in the effect chain for global warming.

In December 1991 at an LCA Workshop in Leiden, a first discussion took place about environmental problems that should preferably be included in an LCA[5,6]. During this workshop an as complete as possible list of generally recognized environmental problems was divided into three groups: depletion including all problem types related to inputs from the environment (extractions), pollution including all problem types related to outputs to the environment (all kinds of emissions), and disturbances including all problem types causing changes of structure within the environment (without associated inputs or outputs). This list is adopted here with some small changes and supplements, see *Table 1*.

Contrary to common practice, three problem types are deliberately left out of this list: space consumption, energy consumption and final solid waste. Space consumption is not included in the list, although in the end the total amount of space is of course limited and might therefore be classified under depletion of resources. However, this problem is rather a physical planning problem than an environmental problem. Much more important than the total amount of space used, is the quality of the space use in terms of degradation of ecosystems. It is proposed here, to classify this aspect under the heading 'physical ecosystem degradation'. Energy consumption is no environmental problem as such but may contribute to a number of problems including resource depletion (including both biotic and abiotic energy carriers), global warming, acidification, nutrification and some disturbances. The same holds true for final solid waste. Final solid waste is not a problem as such, but rather an economic process ('storage of solid waste') causing emissions to water, air and soil, consuming space and producing methane as a potential energy source.

Table 1 Generally recognized environmental problems

Depletion	Pollution	Disturbances
Abiotic resources	Ozone depletion	Desiccation
Biotic resources	Global warming	Physical ecosystem degradation
	Photochemical oxidant formation	Landscape degradation
	Acidification	Direct human victims
	Human toxicity	
	Ecotoxicity	
	Nutrification	
	Radiation	
	Dispersion of heat	
	Noise	
	Smell	
	Occupational health	

Besides these three well known problem types, some problems are left out of the list because it seems difficult to attribute these problems to the functioning of products, such as 'fragmentation of nature areas' and 'depletion of the gene pool', or because they are not yet (or not anymore) generally recognized as environmental problems, such as 'light waves' which is a local Dutch problem in greenhouse areas and 'salination' which is also a local problem. Thus, the list is probably not complete, but can always be extended if there are obvious reasons to do so. On the other hand, the list can perhaps also be reduced because in future closely related problems, such as acidification and nutrification, might be combined on the basis of a common denominator.

The classification according to the problem types mentioned in *Table 1* results in 18 effect scores. Whether this maximum will also be reached in current case studies depends on the question whether classification factors can be developed for all these problem types and whether all inventory data needed are available. Of course, effect scores of problem types can also be zero if the inventory table does not contain any input or output contributing to that particular problem type. Below, possibilities for classification factors will be discussed for each problem of the three categories of problem types.

Depletion

In studies so far conducted, depletion has not systematically been worked out. In several studies different types of fossil energy are added on the basis of their energy content[12,14,15]. Other resources have not yet been included in the assessment of resource depletion.

How could a more comprehensive assessment of depletion be developed? It may be necessary to make a distinction between abiotic resources such as ores and fossil fuels, and biotic resources such as tropical hardwood, ivory and turtle shells, because of the intrinsic value of biotic resources[17], their source function and their role in the maintenance of the life support system[18,19]. (Human life on earth is only possible when temperature, level of radiation, acidity etc. do not exceed certain boundaries. The environment regulates the conditions of the biosphere to a large extent by keeping the global material cycles going. A growing evidence is found that biotic resources play a crucial role in maintaining these cycles. The total of all processes maintaining conditions for life is referred to as life support system[19].)

With respect to abiotic resources, it can be argued as an economic or an environmental problem. If abiotic resources are considered as an environmental problem, however, a factor might either be based on present stocks of these resources (kg) or on the rate of use (kg per year) in relation to their present stocks, measured as years of supply at current rates of use. As concluded during the Leiden workshop[20], there is no general agreement as to which approach is the more relevant. However, there are two serious drawbacks to both approaches. First, the amount available for extraction is highly dependent on market prices of the resources and on available technology. Secondly, exploration usually has a limited time horizon of one to two decades covering the gestation time between discovery and exploitation. These two arguments together have the consequence that a specific figure on the available amount of a resource will always be disputed widely. Despite these drawbacks, a classification of abiotic resources according to present stocks or to rate of use in relation to present stocks seems the best possibility as long as better methods are lacking. Whether data on stocks (and rates of use) are sufficiently available needs further research.

With respect to biotic resources, only critical resources are considered as far as they are not reproduced by a production process; thus, for instance, forestry is not considered as a depletion problem in terms of biotic resources but treated as a production process with its specific environmental impacts (fertilizers). A factor for biotic resources might also be based either on present stocks or on the rate of use in relation to present stocks. Here, the latter approach seems the more relevant one, because the use of biotic resources in principle is only a problem if the rate of use exceeds the regeneration of that particular resource. The result of such a classification would be expressed in years of supply at current net rates of use. Whether data on regeneration are sufficiently available and reliable has to be investigated.

Pollution

As far as the pollution problems are concerned, the general definitions of the classification factors have to be determined per problem type, the specific values are to be derived per separate substance.

Emissions of some substances can in theory contribute to more than one problem but in practice only to one problem. An example here is sulfur dioxide which can contribute to acidification and to human toxicity, but one molecule cannot contribute to both problems during its lifetime. This phenomenon could be indicated with the term 'parallel effect'. An emission can also have more successive effects in practice. For example, nitrogen oxides can actually contribute to both eutrophication and acidification. Other examples are persistent chemicals such as heavy metals or PCBs which can be toxic to ecosystems first and then, through foodchains, also be toxic for humans. This phenomenon could be indicated with the term (direct) serial effect. A serial effect can also be caused indirectly, e.g. methane. One molecule of methane can contribute to photochemical ozone creation and the ozone created contributes in its turn to global warming which can contribute to stratospheric ozone depletion.

In principle, the difference between parallel and direct or indirect serial effects should be taken into account in the classification. Emissions of substances with parallel effects should preferably be classified on the basis of their actual contributions. This is not yet possible, because we lack the necessary data. For this reason all potential effects of an emission with parallel effects are quantified on the basis of the total quantity emitted. In case of an emission of 2 kg SO_2, for example, the contribution to both acidification and human toxicity of the full 2 kg are quantified. This may lead to some double counting. If estimates are available about the average contribution to different problems, this should be taken into account in the classification. Emissions of substances with (in)direct serial effects should in principle be fully classified to all problems concerned. For emissions with indirect serial effects this is not yet possible. For example, although attempts have been made to quantify the indirect global warming effects of hydrocarbons creating photochemical ozone (CH_4, CO, NO_x and non-methane hydrocarbons (NMHC)), the uncertainties about these indirect global warming effects are still too large to use these values[21]. For other indirect serial effects similar attempts have not yet been made at all. Finnveden *et al.*[8] suggest that substances with indirect serial effects should be included separately in the classification and that the total emissions of each substance should be listed (in kg), until indirect classification factors for these substances are available. They also suggest that substances whose classification factors are unavailable but which are known to contribute to a given problem (e.g. NO_x and NO which can have ozone depleting effects and NO_x which plays a role in the formation of photochemical ozone) be included in subscores in the same way. This would, however, result in a large number of subscores with widely varying status.

For example, for the greenhouse effect we could draw up five subscores: one for substances for which the global warming effect can be quantified by means of so-called global warming potentials (in kg CO_2 equivalents), one for the total CH_4 emissions (in kg CH_4), one for the total CO emissions (in kg CO), one for the total NO_x emissions (in kg NO_x) and one for the total emissions of non-methane hydrocarbons (in kg NMHC). In the same way there would be five subscores for ozone depletion and two subscores for photochemical oxidant formation. Hence it would appear to be better and more practical to deal with these uncertainties as such. For example, flags (qualitative remarks) could be attached to substance emissions which may have indirect effects. The values of the associated indirect GWPs and ODPs at which the outcome of the LCA would change could then be calculated in a reliability analysis (see Valuation section). It could then be considered whether the values calculated are realistic, given the current level of understanding. Substances known to contribute to certain problems but for which classification factors cannot yet be determined could be dealt with in a similar manner.

Another question is how to deal with spatial differentiation in the development of these factors. Spatial differentiation can be relevant if for example the degradation of and the sensitivity for a substance differs per type of soil. Then, two approaches may be followed. A more generic approach would be to define these factors for different relevant media, and to specify the average surface of the media in the given study area. A more site-specific approach would be to localize the relevant media on a map and to relate their distribution to the emission dispersion and deposition pattern that then should be given as well. Both approaches are in principle possible at all scale levels (global, continental, regional, local).

To give an example, we may regard the effects of deposition of acid rain in relation to the geographical distribution of sensitive, non-buffered areas in Europe. This is done in the acidification model RAINS developed at the International Institute of Applied Systems Analysis[22]. Some authors seem to argue that such a site-specific approach, including the factual spatial distribution, should be aimed at in LCA[23,24]. However, such an approach sets extremely high demands on data in both the inventory and the classification. The inventory should include a geographical specification per economic process and the classification should include data on geographical distribution of relevant media per type of problem and dispersion and deposition patterns per chemical. The site-specific approach is therefore not generally feasible. For this reason the first approach, specification of media in averages per spatial level, seems to be preferable at this time. If necessary, it might, for example, be possible to divide the world in ten regions for which regionally differing factors might be determined. If these regions can be the same for each problem type, the data increase could remain possible to survey.

A related question is how to deal with problems which are caused by a combination of emissions, such as nutrification. This problem is caused by emissions of nitrogen and phosphate but on a specific site only one of these can be in the minimum causing the actual effect. In a generic classification we propose to classify both emissions to their potential effect and leave out the site-specific differences.

In a recent study[25] we made a first elaboration of classification factors for a generic classification at a global scale. This means that the classification factors are not differentiated to different areas. For example, there will be worked with one 'global average soil composition' or one 'globally representative soil composition'. Below, generic classification factors will be discussed per problem type at this global scale. The factors proceed from our recent study[25] in which a number of suggestions made by Finnveden *et al.*[8] are included. The factors will be discussed in the sequence as listed in *Table 1*.

For ozone depletion so-called ozone depletion potentials (ODP)[26,27] can be applied as classification factors. The ozone depletion potential of a gas is based on models simulating relevant environmental processes, which play a role in ozone depletion, and it is defined relative to a reference substance, in this case CFC-11.

For global warming the above mentioned global warming potentials (GWP)[21,28] can be applied as classification factors. The GWP is derived in a similar way as the ODP, but defined relative to CO_2.

For photochemical ozone formation so-called photochemical ozone creation potentials (POCP) are being developed[29,30]. POCPs are only defined for volatile organic compounds (VOCs). However, the results of this research are still the subject of discussion, and it is not yet clear whether they are appropriate[25].

Acidifying emissions can be classified based on the potential number of H^+-equivalents they can form. This method has been used in a number of studies [14,15]. Heijungs *et al.*[25] define the classification factors for acidifying emissions in terms of an acidification potential (AP) relative to SO_2.

The definition of appropriate classification factors for human toxicity and ecotoxicity is one of the main methodological bottlenecks of the classification. As mentioned, the critical volumes approach has been the practice up till now. In this approach an exposure analysis is lacking and the effect analysis is based on semi-political standards. It is possible to improve this current practice by developing the classification factors for human toxicity and ecotoxicity along two parts: an exposure part relating the emissions to a concentration to which a receptor can be exposed, and an effect part relating this exposure to the effects on a human being or ecosystem. In principle, these two parts (exposure and effect) are also the basis of classification factors of other problem types, although they are often difficult to distinguish. Both exposure and effect parts will have to be determined for each substance and as much as possible based on scientific models and empirical data.

For both human toxicity and ecotoxicity exposure could be calculated with the multimedium environmental models of Mackay[31]. However, Mackay models cannot be applied directly to emissions as quantified in LCAs because LCA-emissions are not restricted to a certain period of time. Emissions of a substance during the product's entire life cycle take place at a non-homogeneous and unknown rate. An LCA is only concerned with the total emission of a substance associated with the entire life cycle of a product, which is regarded as a pulse (in kg). Multimedium environmental models, which take into account time-dependent processes such as degradation and partitioning, are necessarily based on a flux (e.g. kg day^{-1}). There is a relation between the flux and the equilibrium concentration. Increasing the flux leads to an increased concentration, and thus to an increased risk. A solution for this flux-pulse problem can be found by selecting a reference substance and calculating a dimensionless classification factor per substance similar to the ODP-, GWP- and POCP-concepts[32].

In defining the effect part for human toxicity, a solution has to be found for how to deal with the large amount of mechanisms and effects involved and for how to deal with for example carcinogenic substances, for which it is impossible to define a threshold value. We suggest that we solve these problems by taking human beings as the endpoint of the classification, apply threshold values for the first occurring adverse effect and derive 'virtual' threshold values for non-threshold substances by defining tolerable (thus not purely scientific) levels of an increased risk on cancer. In this way, a so-called HTP (human toxicity potential) may be developed for each substance[32]. Such an HTP indicates the human toxicity of a particular emission of a substance relative to the human toxicity of an equal emission of a reference substance.

In defining the effect part for ecotoxicity, the same problems as mentioned for human toxicity have to be solved. In addition, an assessment of ecotoxic effects has to take into account the large number of species within an ecosystem. A solution for these problems may be found along the same line as for human toxicity, except that threshold values for ecosystems have to be derived from a number of single species toxicity data. For this several methods have been developed which may be applied to derive these threshold values[33–35]. We propose to distinguish between terrestrial and aquatic ecosystems, because of the different species present in these media and the different exposure routes of these species. As specific toxicity data for ecosystems in the sediment and for exposure of ecosystems to air are lacking, these compartments are not yet considered.

In this way a so-called TETP (terrestrial ecotoxicity potential) and an AETP (aquatic ecotoxicity potential) may be developed for each substance[32]. The TETP and the AETP indicate the toxicity for terrestrial and aquatic ecosystems, respectively, due to a particular emission of a substance relative to the toxicity for terrestrial and aquatic ecosystems of an equal emission of a reference substance. For further details about the HTP-, TETP- and AETP-proposals, see Guinée and Heijungs[32]. It must be stressed here that the HTP-, TETP- and AETP-approaches are still in an early stage of development and that concrete values have not yet been derived for any substance.

For nutrification, an assessment of nitrogen and phosphorus might be based on their average presence in biomass (approximately 7 to 1 kg). It has to be considered that then clearly potential effects are added up because, in line with the Law of Liebig, in practical situations only the nutrient which is in the minimum will have an effect. Aquatic emissions of organic material, usually measured as the chemical oxygen demand (COD) can be included in the classification of nutrifying substances. Finnveden *et al.*[8] and Heijungs *et al.*[25] have developed different proposals for this. Finnveden *et al.* suggest a separate definition of scores for aquatic and terrestrial nutrification, and to express aquatic nutrification in terms of COD and terrestrial nutrification in terms of nitrogen equivalents (kg). Heijungs *et al.* suggest that the potential creation of biomass is taken as endpoint of the classification and define one encompassing score for aquatic and terrestrial nutrification. They propose to define the classification factors for eutroficating emissions in terms of a nutrification potential (NP) relative to phosphate.

Other pollution problems on the list of problem types are radiation, dispersion of heat, noise, smell and occupational health. For the classification of radiation emissions the critical volumes approach may be applied as long as better methods are lacking. An emission of a potentially radioactive substance is then divided by its radiation threshold value for occupational health. The International Commission on Radiological Protection has defined an annual limit of intake for this[36].

Dispersion of heat is only a substantial environmental problem in aquatic ecosystems. It may be expressed in standard energy units (joules) emitted to water which can directly be derived from the inventory.

Noise is usually expressed in decibels. However, decibels cannot be added 1 to 1. To enable such an addition and a linear allocation to a unit of output produced by an economic process, decibels could be converted to Pa^2 yr[25]. In fact, this conversion is a subject of the inventory. These inventory noise data can be added without further assessment in terms of 'potential noise'. Then, the classification factor is one for all types of noise.

For the classification of smell, again the critical volumes approach may be applied as long as better methods are lacking. An emission of a potentially odorous substance is then divided by its odour threshold value[37].

For occupational health, factors are not yet developed. This problem type can be subdivided into human toxicity, radiation, noise, smell and victims within the internal environment of a factory. Separate classification factors may be developed for each of these internal occupational problem types analogous to the factors defined for the same problem types in the external environment[25].

Disturbances

As mentioned before, disturbances are generally not easy to relate to the functioning of product systems. Desiccation due to water extractions has not been considered in any study to this day, but it may be expressed in terms of water use (kg) in a generic classification without any further spatial differentiation. In a spatially differentiated classification it could be expressed as the ratio of water use and local or regional water stocks. The latter has certainly a more direct relation to desiccation, but has to deal with the mentioned drawbacks associated with spatial differentiation.

Ecosystem degradation has to our knowledge been treated in only one study. The use of the resource hardwood from tropical rainforests has been assessed as use of scarce renewable resources (kg) and in terms of the surface of degraded ecosystems (ha)[15]. Frischknecht[38] and Finnveden[9] have developed a method for ecosystem degradation based on five categories (natural systems, modified systems, cultivated systems, built systems, degraded systems) of ecosystems defined by the IUCN/WWF/UNEP[39]. Further research is needed to make this interesting suggestion practicable[25].

Landscape degradation is partially included in the ecosystems categories of the IUCN. The development of a separate factor for this problem seems to be very difficult.

Human victims as a direct consequence of the (dis)functioning of a process, might be regarded as an environmental problem. If in the inventory slightly, seriously and fatally injured are distinguished, a further classification would be necessary. As far as we know there are no methods (yet), which could be applied for this classification. If only data on fatally injured are known or if there is a fixed relation between the number of fatally injured and the number of slightly and seriously injured, human victims could be expressed in a number of fatal casualties[25]. This number can directly be derived from the inventory.

Concluding, it seems possible to define classification factors for quite a broad spectrum of problem types. However, all factors need further improvement and continuous updating. To coordinate and authorize this process, it is vital to have a scientific discussion panel for each of these problem types, such as the Scientific Assessment Panel under the auspices of the World Meteorological Organization (WMO) for ozone depletion, the Intergovernmental Panel on Climate Change (IPCC) under the auspices of WMO and United Nations Environment Programme (UNEP) for global warming, and the Working Group on Volatile Organic Compounds under the auspices of United Nations Economic Commission for Europe. For other problem types such panels are still lacking.

Multiplication and aggregation

Third step of the classification is the multiplication of environmental inputs and outputs with their specific classification factors and the aggregation of the results per problem type. It is suggested to call the result of this conversion the environmental profile of a product [12,14,15]. For example, each emission of a potential greenhouse gas, for which a GWP-value is available, is multiplied with its specific GWP-value as published in the IPCC scientific assessment report[28]. As the global warming potential of a gas is defined relative to carbon dioxide, the result of each multiplication can be expressed in mass equivalents of carbon dioxide. These CO_2-equivalents can be added which results in one overall score for global warming.

An optional in between step is the conversion for individual processes or groups of processes. The result of this step is called the environmental profile of a process or group of processes. These may be useful in the identification of improvement options; see below.

Normalization of effect scores

Final step of the classification is the normalization of effect scores. The effect scores obtained after the previous three steps denote the contributions to well-known environmental problems. The meaning of the resulting numbers, however, is far from obvious. The effect scores become more meaningful by converting them to a relative contribution to the different problem types by means of a normalization[25,40]. To this end, we propose to divide the effect scores by the total extent of the relevant effect scores for a certain area and a certain period of time. The result of this step may be called the normalized environmental profile. All normalized effect scores have the same dimension: that of a time.

The total extent should be calculated using empirical data about extractions and emissions, and applying the classification models proposed above. Since these are generic classification models at a global scale, data on extractions and emissions for the normalization should be gathered on a global scale for a certain time period, for example a year. The global extents of effect scores can probably be estimated for depletion of abiotic resources[41,42], ozone depletion and global warming[28]. For the other problem types, data have still to be gathered.

Valuation

The fourth component of the environmental LCA deals with the final environmental problem appraisal based on the environmental profile(s) of the product(s) studied, taking into account the reliability and validity of the results by performing sensitivity analyses. Thus, two elements may be distinguished here: valuation of the effect scores of the environmental profiles; and assessment of the reliability and the validity of the results.

Valuation of the effect scores of the environmental profiles

The classification will result in an environmental profile, which as much as possible is still the product of empirical knowledge about economic and environmental processes. Thereafter, a valuation can be desirable for both product comparison and product improvement. In product comparison the effect scores of the environmental profiles of different products often have to be weighted in relation to each other, while for product improvement a weighting of the effect scores of the product under study is necessary to determine on which aspects the product should be improved primarily. In principle, social values and preferences dominate in this valuation. These values and preferences could be approximated with policy aims, costs, or with the help of experts or an expert panel and then be the input of a qualitative or quantitative multicriterion analysis. Policy aims and/or costs have proven to be practical indicators in methods which treat the classification and the valuation as a unitary methodological component[2,4,10]. It could be further investigated whether policy aims and/or costs are also appropriate and practical indicators in a separate valuation.

Here, we will focus on qualitative and quantitative multicriterion analyses which make use of experts or expert panels. However, first it should be determined whether such a weighting is necessary at all. This means that there is a check on whether one alternative is better than or equal to all other alternatives on all criteria. If so, the outcome is clear without further weighting. In some studies on milk packaging, for example, a PE-milkbag compared to glass and carton packages[12,43], and a polycarbonate milk bottle compared to glass and carton packages[14] scored equal or better on all criteria considered. If this unweighted comparison does not lead to a result, as will often be the case, and one aims at a conclusive result, a qualitative or a quantitative multicriterion analysis can be performed[44].

In a qualitative multicriterion analysis effect scores are weighted against each other in a non-formalized way. This means that for each separate case study the weighting is performed by an individual expert or by a panel of experts. For major decisions such as the granting of ecolabels, the establishment of a panel seems preferable if representing the relevant scientific and social opinions. Moreover, a judgement is almost always possible and qualitative aspects can easily be included. This method is followed by several countries in their ecolabelling systems. Thus, in the German ecolabelling system a group of experts gives their judgement based on the information offered to them. In Canada the ecolabel is based on a combined decision by a government body and the private Standards Association. Disadvantage of the qualitative multicriterion analysis based on a panel is that it does not seem a workable option for more daily applications such as product improvement and development within companies.

In a quantitative multicriterion analysis effect scores are weighted in a formalized way. This means that the weighting is performed according to a formula applying a list of weighting factors. The effect scores are multiplied with the corresponding weighting factors and the results of this multiplication are aggregated into one so-called environmental index. The disadvantages of the quantitative multicriterion analyses are that qualitative aspects are difficult to include and that the environmental index suggests a scientific precision which cannot hold true. An important advantage is the reproducibility of the results. The weighting factors can be determined per case study or, in a more generic way, for all case studies for a certain period of time, for example a year. The advantages of a quantitative multicriterion analysis increase if it is based on such a standard list of weighting factors, because the costs can be reduced substantially and the method is easily applied. These latter aspects are very important in a society which produces and consumes products every day. The main problem in elaborating such a standardized quantitative multicriterion analysis, however, is the definition of the weighting factors with a sufficiently broad social basis. Further consideration should be given to this point.

Evaluation of the reliability and the validity of the results

A valuation of environmental profiles without an assessment of the reliability and the validity of the results, is of little value. The step concerns a sensitivity analysis regarding the influence of both the uncertainty of data and the assumptions and choices made.

The reliability of the results can be assessed using various techniques[45]. Classical error analysis yields results with a margin of uncertainty (e.g. 10 ± 2), provided that (some of) the data (process data, classification factors, etc.) are specified in this form. For the data for which no margins of uncertainty are specified or estimated, a so-called marginal analysis can be performed, indicating the process data which should be known most accurately, because they have a crucial impact on the results of the particular study. The marginal analysis is a mathematical tool[45], which reveals the sensitivity of the result as a function of small changes of the process data. As a consequence, the results of the marginal analysis can also be used for the improvement analysis: see below.

For an assessment of the validity of the results, there is as yet no such systematic treatment. Assumptions and choices underlying the methodology and the particular case study influence the results of the study. The specifications of the products considered, the allocation rules for multiple processes, the environmental problems considered and the composition of an expert panel for valuation are examples of choices and assumptions in each one of the previous components of an LCA.

In the studies so far little attention has been paid to the assessment of the reliability and the validity of the results. However, for the credibility of LCA-studies, it is very important that these aspects receive much more attention. LCA-researchers have to face the problem of the influence of unreliable and unknown data on and the limitations of their results. In addition to the sensitivity techniques discussed above, peer reviews could of course also discuss and thereby increase the reliability and validity of the results.

Improvement analysis

To date, improvement of products was undertaken by designers on a trial-and-error basis using empirical knowledge on environmental properties of materials and processes. The improvement analysis of an LCA can structure this process. Combined with expertise in other fields, such as costs and technological feasibility, the improvement analysis may yield options for the redesign of a product. One of the applications of an LCA is the product improvement itself: the options from the environmental analysis lead after an exhaustive evaluation including all relevant aspects (environmental, financial, convenience, safety, etc.) to a new product.

In a recent paper[45], a methodological aspect of the improvement analysis has been worked out in two complementary methods: the dominance analysis and the marginal analysis.

In the dominance analysis, the main origins of the environmental problems are traced back. The inventory tables per process may be very useful in finding the options for improvement, because substances or groups of substances that are considered as a major problem can be traced back to processes or groups of processes responsible for those bad scores.

For the improvement of products, knowledge of the dynamic behaviour of the environmental profile in terms of process modifications can be even more important. The marginal analysis is a technique which addresses this question. Processes to improve can be preselected using knowledge of the sensitivity of the result (e.g. impact table or environmental profile) to small peturbations in the economic or environmental process data. A designer or process technologist can thus be informed about the best starting points for product improvements. As mentioned before, procedures for this are currently being worked out[45].

With mathematical procedures for the identification of improvement options and the inclusion of expertise from process technologists and designers, LCA might become an analytic tool for eco-design supporting a continuous environmental improvement of products.

Summary and conclusions

Three components of the methodological LCA-framework have been discussed: the classification, the valuation and the improvement analysis. In a previous article the other two components, the goal definition and the inventory, were discussed.

The aim of the classification is to quantify per problem type the contribution of environmental inputs and outputs of a product system to a number of generally recognized environmental problems. The result can be called the environmental profile consisting of a number of effect scores. In the development of such a classification four steps can be distinguished: 1. the definition of generally recognized environmental problem types which should be considered in an LCA; 2. the definition of classification factors indicating the contribution of one unit of an environmental input or output to a particular environmental problem; 3. the multiplication of environmental inputs and outputs with their classification factors and subsequent aggregation of the results per problem type into a number of effect scores; and 4. the normalization of the effect scores.

A list of 18 problem types is given, subdivided into three main categories: depletion, pollution and disturbances. Whether all these problem types can be included in a case study depends on the availability of inventory data and classification factors. With respect to the classification factors it appears that for most pollution and disturbance problems factors can be defined, although they need further improvement; for the depletion problems these factors are still missing. Besides new development or improvement, all classification factors will need continuous updating. These activities should preferably be coordinated by specialized scientific fora. To get a better indication of the meaning of the different effect scores, they could be divided by the total extent of the environmental problems considered.

The valuation component consists of a valuation of the effect scores of the environmental profile and an assessment of the reliability and validity of the results. In principle, social values and preferences dominate in the valuation of the effect scores. It should first be checked whether one alternative is better than or equal to all other alternatives on all criteria. If so, a further valuation is not necessary. If such an unweighted comparison does not lead to a result, as will often be the case, and one aims at a conclusive result, a qualitative multicriterion analysis or a quantitative multicriterion analysis could be performed making use of experts, expert panels, or a standard list of weighting factors. A quantitative multicriterion analysis based on a standard list is an easily applicable method and therefore seems the most preferable. However, the main problem here is the compilation of such a standard list, with a sufficiently broad societal basis.

A valuation of environmental profiles without an assessment of the reliability and the validity of the results, is of little value. Methods for this are currently being worked out. In studies so far little attention is paid to the assessment of the reliability and the validity of the results. With respect to the credibility of LCA-studies, it is very important that these aspects get more attention.

In the improvement analysis possible improvement options are identified. For this, two complementary analysis techniques can be applied: the dominance analysis and the marginal analysis. With these two types of analyses, a number of options can be generated to improve a particular product. For the assessment of the feasibility of these options, other expertise, outside the field of LCA, is necessary. It is concluded that with mathematical procedures for the identification of improvement options and the inclusion of expertise from process technologists and designers, LCA might become an analytic tool for eco-design supporting a continuous environmental improvement of products.

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