A Critical Review of Operational Valuation/Weighting Methods for Life Cycle Assessment

Survey

Göran Finnveden
FMS, Forskningsgruppen för miljöstrategiska studier, Stockholms Universitet/Systemekologi och FOA

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Abstract

There are currently several methods available and in use for valuation (weighting) within the framework of Life Cycle Assessment. In case studies they often produce different results. The aim of this paper is to critically review presently available and operational methods in order to see if any of them can be recommended for use. Among the methods that are reviewed is the EPS-system, the Tellus method, economic valuation based on the impact pathway analysis approach, the ecoscarcity method, the Eco-indicator 95 and other distance to target methods. The major conclusion is that none of the presently available methods can be recommended for use as an LCA valuation (weighting) method today. This is mainly because they either suffer from significant datagaps, include inconsistencies, lack justification of major assumptions, or because they are not valuation methods at all. Better methods can however be developed which fulfil basic requirements concerning consistency, justification of assumptions, transparency and without to severe datagaps. Further research in this area is therefore needed and motivated. Research should focus on monetisation and panel methods. Other approaches have inherent limitations which limit their usefulness as weighting methods.

Table of Contents

Abstract i
Table of Contents ii
Preface iii

1 Introduction 1
1.1 Life Cycle Assessment 1
1.2 The framework 1
1.3 Weighting methods 6
1.4 Aim of this paper 7
2 Quantitative, operational weighting methods 9
2.1 Introduction 9
2.2 Proxy approaches 10
2.3 Technology 10
2.4 Monetisation methods 11
2.4.1 Introduction 11
2.4.2 The EPS-system 15
2.4.3 The Tellus system 20
2.4.4 Economic Valuation Analysis (EVA)/Impact Pathway Analysis 20
2.4.5 Other approaches 22
2.4.6 Non-reviewed approach 23
2.5 Authorised targets/standards (distance to target methods) 23
2.6 Panel methods 28
3 Qualitative methods 32
4 Conclusions and Discussion 35
References 39

Preface

This report is an update and a further elaboration of parts of an earlier study (Finnveden, 1996). However, it has more far-reaching conclusions.

I am grateful to number of people who have helped me either by providing me with literature or by sending written comments on an earlier draft of this report, or both. A big Thank You to Andreas Ciroth (Technical University, Berlin), Philippa Dobson (PIRA, Leatherhead), Mark Goedkoop (PRé, Amersfoort), Michael Hauschild (IPU, Copenhagen), Patrick Hofstetter (ETH, Zürich), Gjalt Huppes (CML, Leiden), Mattias Höjer (fms, Stockholm), Jessica Johansson (fms, Stockholm), Wolfram Krewitt (IER, Stuttgart), Erwin Lindeijer (IVAM-ER, Amsterdam), Åsa Moberg (fms, Stockholm), Frank Oosterhuis (IVM, Amsterdam), Viveka Palm (Statistics Sweden, Stockholm) and Bengt Steen (Chalmers, Gothenburg). Discussions within the Task Group on Normalisation and Weighting working under the SETAC-Europe Working Group on Life Cycle Impact Assessment have also been useful. The responsibility for the text and the conclusions is of course with the author.

1 Introduction

1.1 Life Cycle Assessment

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, 1997a). The general categories of environmental impacts needing consideration include resource use, human health, and ecological considerations (ibid.).

LCA is one tool used for describing environmental impacts. Examples of other environmental management tools include Risk Assessment, Environmental Impact Assessment, Environmental Auditing, Substance Flow Analysis, Energy Analysis, and Material Flow Analysis (see e.g. Anonymous, 1997, Cowell et al., 1997, Moberg, 1999, Moberg et al, 1999, Wrisberg and Gameson, 1998, for a discussion on related tools). What makes LCA unique is the "cradle-to-grave" approach combined with its focus on products, or rather the functions that products provide. "Products" are interpreted in a "broad" sense, and include both material products and services. For example, "taking care of municipal solid waste" is a product that may be studied by LCA.

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, 1997a). LCA is increasingly used by companies (Baumann, 1996; Berkhout and Howes, 1997; and Grotz and Scholl, 1996), and government agencies (Curran, 1997).

1.2 The framework

This section gives a brief presentation of the LCA framework and terminology, based primarily on ISO (1997a). Figure 1 illustrates the framework and also some applications of LCA, which are outside the LCA framework.

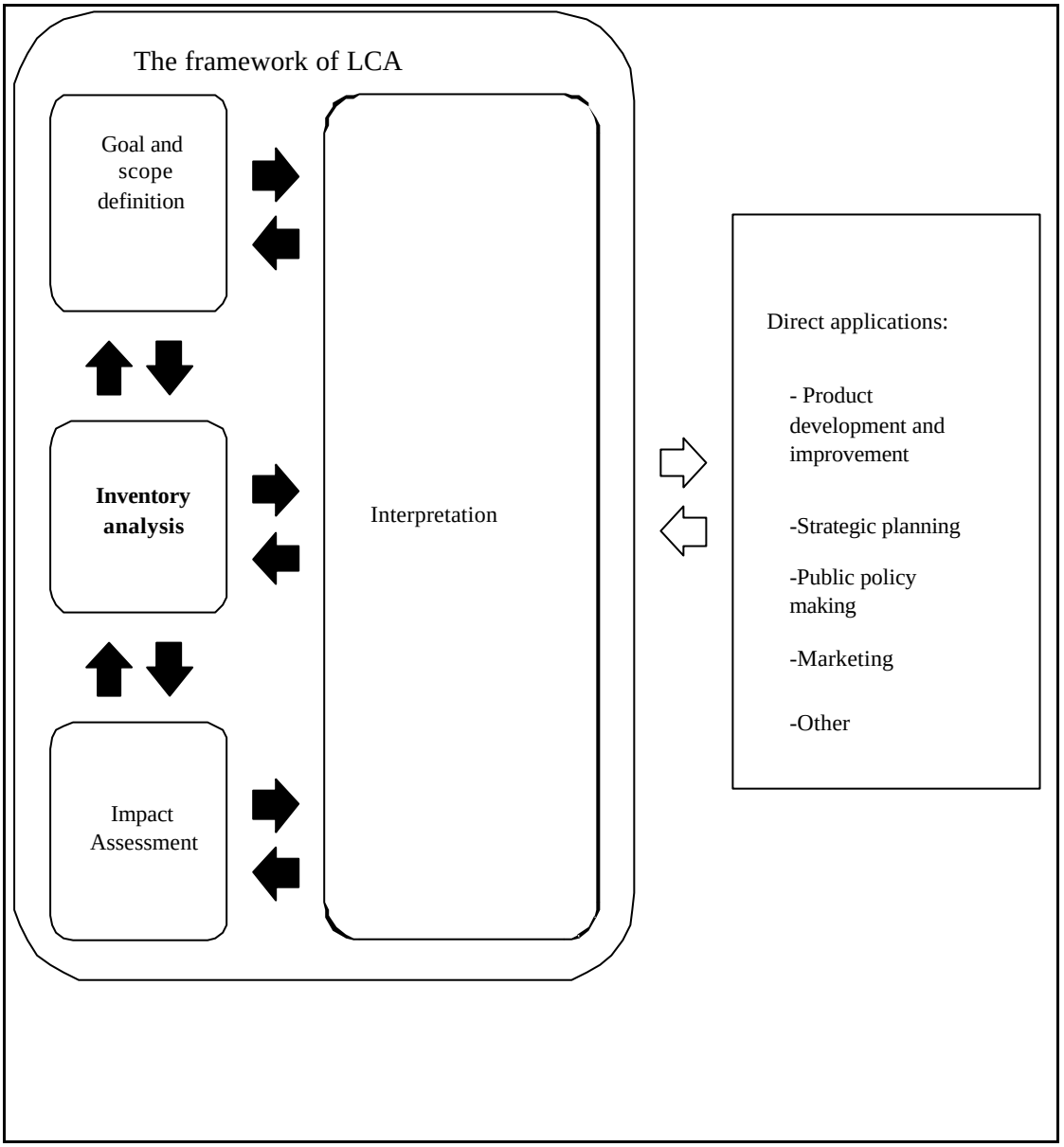

Figure 1. Phases of an LCA (ISO, 1997a).

A life-cycle assessment includes four phases (ISO, 1997a):

1. Goal and scope definition.

2. Inventory analysis, involving the compilation and quantification of inputs and outputs for a given product system throughout its life cycle. The inventory analysis results in a large table of all inputs to the system (resources, etc.) and all outputs from the system (emissions, etc).

3. Life-cycle impact assessment, aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts of a product system (ISO, 1997a). This phase may include elements such as:
3.1. Classification (in which different inputs and outputs are assigned to different impact categories based on the expected types of impacts to the environment).
3.2. Characterisation (relative contributions of each input and output to its assigned impact categories are assessed and the contributions are aggregated within the impact categories).
3.3. Valuation (weighting across impact categories).
4. Interpretation, in which the findings of either the inventory analysis or the impact assessment, or both, are combined in line with the defined goal and scope.

There is a separate standard for the goal definition and inventory analysis (ISO, 1998a) and standards for life-cycle impact assessment (1998b) and life cycle interpretation (ISO, 1997b) are in preparation. The draft international standard for life cycle impact assessment (LCIA) contains some changes in framework and terminology compared to the standard on principles and framework. The draft LCIA standard contains a number of mandatory and optional elements. The mandatory elements are:

1. Selection of impact categories, category indicators and models.
2. Assignment of LCI results (Classification)
3. Calculation of category indicator results (Characterisation)

The optional elements are:

1. Calculating the magnitude of category indicators results relative to reference value(s) (Normalisation)
2. Grouping
3. Weighting
4. Data quality analysis (mandatory element in a comparative assertion).

It can be noted that “valuation” is now called “weighting”. In this paper, these two terms will be used interchangeably. It can also be noted that weighting is regarded as an optional element. There are also other changes in terminology and framework compared to the “old” framework (ISO, 1997a), but these have only a limited influence on the discussion here.

One important choice when defining the scope of an LCA is the type of impacts included. This is also a part of the first mandatory element of the LCIA according to the draft ISO standard (ISO, 1998b). According to the LCA definition, a study should include “the environmental aspects and potential impacts” (ISO, 1997a), suggesting implicitly that all relevant environmental aspects should be included. Different, although not identical, lists of impact categories have been suggested. The “check-list” from the Nordic Guidelines (Lindfors et al, 1995a) is presented in Table 1. Another example that could have been chosen is the default list suggested by Udo de Haes (1996).

Table 1. List of impact categories, the categories can be further divided into subcategories (Lindfors et al, 1995a).

Impact category
1[']. Resources - Energy and materials
2. Resources - Water
3. Resources - Land (including wetlands)
4["]. Human health - Toxicological impacts (excluding work environment)
5["]. Human health - Non-toxicological impacts (excluding work environment)
6["]. Human health impacts in work environment
7. Global warming
8. Depletion of stratospheric ozone
9. Acidification
10. Eutrophication
11. Photo-oxidant formation
12. Ecotoxicological impacts
13["']. Habitat alterations and impacts on biological diversity
14["""]. Inflows which are not traced back to the system boundary between the technical system and nature.
15["""]. Outflows which are not followed to the system boundary between the technical system and nature.
' This impact category can be divided into several subcategories, e.g., a division can be made between energy and materials, and/or between renewable and non-renewable resources. These choices can be made in relation to the choice of characterisation methods.
" Work environment is one among other exposure situations for humans. The suggestion to treat this exposure situation separately is partly due to available characterisation methods.
"'Several of the impact categories can as a second order effect cause "Habitat alterations and impacts on the biological diversity". This impact category is however related to activities and emissions which can have a direct impact.
""" Not impact categories but should be included.

An important feature of the ISO framework is the Interpretation phase where overall conclusions from the study are to be drawn. These conclusions should be based on information from all other phases and elements of the LCA. This implies that the results from the valuation element will not necessarily be identical to the conclusions drawn in the interpretation. Using a valuation method is thus something different from drawing conclusions (Lindfors et al, 1995a). To use a valuation method can be a way of analysing the results. Results from the valuation can be conditional: if a certain valuation method with a certain set of values is used then a certain result is obtained. By using several valuation methods and sets of values, several results can be obtained, which can be used when conclusions are to be drawn and recommendations are to be made in the interpretation phase.

The framework described above has been developed during the 1990s. It is largely the same as in SETAC's 'Code of Practise' (Consoli et al, 1993), and in different guidelines (e.g. Heijungs et al., 1992; Vigon et al., 1993; Lindfors et al., 1995a; and Wenzel et al., 1997). The development of impact assessment methods during the 1990s has in principle followed two different paths, development of single-step methods and multi-step methods (Guinée, 1994). The single-step methods do not separate between the different subcomponents of the impact assessment. Examples are the Ecoscarcity method (Ahbe et al, 1990) and the EPS-system (Steen and Ryding, 1992). In the development of these methods, the starting point was the decision that a one-dimensional score is needed. From this viewpoint there was no need to perform the impact assessment in several steps. (It can however be noted that it has later been suggested that both methods can be developed in order to fit in the SETAC-framework (Müller-Wenk, 1994, Steen and Ryding, 1994)).

The multi-step methods on the other hand were developed largely based on the wish to separate, as far as possible steps based on environmental sciences from steps based on ethical and ideological valuations. From this starting point, a separation into several elements was relevant. Examples of projects which followed this path was the development of the Dutch guidelines (Heijungs et al, 1992), the 'LCA-Nordic'- project (Anonymous, 1992, Lindfors et al, 1995a) and the EDIP-project (Wenzel et al, 1997). In these projects, the emphasis was placed on the classification and characterisation methods, while the valuation methods were given much less attention. Only lately has attempts been made to develop valuation methods based on the multi-step procedure. It can therefore be argued that it is still an open question to what extent the framework described above is useful when the aim is to develop valuation methods (Hofstetter, 1998).

It is generally recognised that the valuation element requires political, ideological and/or ethical values and these are influenced by perceptions and worldviews. Not only the valuation weighting factors, but also the choice of valuation methodology, and the choice of using a valuation method at all, are

influenced by fundamental ethical and ideological valuations (Finnveden, 1997). Since there is presently no societal consensus on some of these fundamental values, there is presently no reason to expect consensus either on valuation weighting factors, or on the valuation method, or even on the choice of using a valuation method at all. If no valuation method is used at all, comparisons are made category by category, and not on an aggregated level.

Even if the preceding phases and elements are mainly based on more or less traditional natural sciences, this should not be interpreted as if they are free from value choices. Some fundamental values can have repercussions on methodological choices for the inventory analysis, and the classification and characterisation elements (Finnveden, 1998, Hofstetter, 1998). This is also illustrated in the Eco-indicator 98 project where not only the valuation weighting factors, but also some methodological choices for other elements in the impact assessment are determined by value choices (Goedkoop et al, 1998).

1.3 Weighting methods

Several methods for valuation in connection with LCA has been developed during the 1990s and are being used. Several reviews and discussion papers on valuation methods have been published during the last years (e.g. Bengtsson, 1998, Braunschweig et al, 1994, Eriksson et al, 1996, Finnveden, 1996, Giegrich et al, 1995, Hertwich et al, 1997, Hofstetter, 1996, Lindeijer, 1996, Lindfors et al, 1995a and b, Magnussen et al, 1998, and Powell et al, 1997). It has also been shown that different valuation methods in case studies will give different results (e.g. Baumann and Rydberg, 1994, Lindfors et al, 1995a and b, Braunschweig, 1994, Guinée, 1995, Magnussen et al, 1998, Notarnicola et al, 1998).

Valuation methods can be classified in different ways. A first distinction can be made between qualitative (including semi-quantitative) and quantitative methods. A second distinction can be made among the quantitative methods between methods for deriving generic sets of weighting factors which can be used in several cases (and thus is case independent), and methods which aim at results which are only to be used in connection with a specific case study (and thus is case-dependent) (Lindeijer, 1996, Miettinen and Hämäläinen, 1997). From a methodological standpoint, this distinction is however of limited importance since essentially the same methods can be used when developing weighting factors both for generic and for case-specific use. The distinction is however relevant from another standpoint. If a case-specific weighting set is used, derived for the specific case, the difference between using valuation methods and drawing conclusions in the interpretation phase may not be so obvious since applying valuation methods is perhaps more or less a way of systematising the process of drawing conclusions. In the case of generic weighting sets, the distinction between using valuation methods and drawing

conclusions may be more important because those that are to draw conclusions may regard the valuation methods as more or less authoritative and reliable.

All quantitative methods discussed in this paper results in valuation weighting factors, V_i, which expresses the contribution to the total potential environmental impact from the intervention or impact category i. The total potential environmental impact, EI, can then be calculated as

$$EI = V_i I_i$$

where I_i is the score for either the intervention i or the impact category i.

The methods discussed here thus only considers a linear combination between interventions (or impact scores) and weighting factors. Other types of equations could in principle be discussed (Heijungs, 1994). However, so far a linear combination has been assumed in presently available methods.

1.4 Aim of this paper

There are several valuation methods that are currently being used in LCA. They may in specific case studies give different results. A question often posed is therefore: Which valuation method (or methods) should be used in a case study? Which methods are reliable? The aim of this report is to critically review presently available and operational methods in order to see if any of them can be recommended for use. Due to the questions asked, the focus will be on weaknesses of the reviewed methods, rather than the positive aspects. The focus will be on quantitative methods with generic sets of valuation weighting factors that are ready to use. Qualitative methods and other approaches that do not have published generic sets of weighting factors will only be briefly reviewed. The methods are reviewed as they appear today. Many of them are under development, and they may change. The potential of different methods is however not reviewed. A secondary aim is that the paper should be possible to use as a starting point for discussion on valuation methods, and for further developments.

The focus in this report is on weighting methods. For methods where single-step valuation weighting factors have been published, it can also be motivated to include characterisation methods which have been used in the review. Other parts of the methodology, such as uncertainty analysis, are however not included here since they are in principle independent from the weighting method.

Can valuation methods be critically reviewed? The question is relevant since it is difficult to critically review values. There are however some other aspects which can be reviewed. For example, most methods include parts that are scientific (including all types of sciences). These scientific parts can then be reviewed. One can also study whether there are any logical contradictions within the method, or whether there are any differences between the description of the method and its operationalisation. Another interesting question concerns which type of environmental impacts are considered. An

evaluation whether the results make sense and to what extent they are in line with common societal perceptions can also be made. The latter is of course very difficult, but not completely impossible.

Different requirements on weighting methods can be formulated and these can be used for evaluating different methods. The list in Table 2 is from the previous SETAC-Europe subgroup on Normalisation and Valuation (Lindeijer, 1996). It is divided into general criteria and possibly contradictory criteria:

Table 2. Requirements for weighting methods (Lindeijer, 1996)

General requirements
1. Reflect the subjective characteristic of weighting in general
2. Fit the purpose of the LCA, including time/money constraints, communication requirements and effectivity for control action
3. An inter-effect weighting should be included somehow
4. Deal with uncertainty
5. Applicable to all present environmental problems
6. Flexibility to include new problems
7. Be explicit/transparent; e.g. be explicit on the weighting criteria used, perform all weighting substeps explicitly and ensure verifiability
8. The units for expressing the impacts should make no difference for weighting
9. Use exactly the same formulation of problem types as in classification and characterisation
Possibly contradictory requirements
10. Keep the weighting principles simple and understandable
11. Include all available natural science information
12. Incorporate general interests of all involved agents
13. Differentiate for regions and time discounting
14. Use single scores or indices for optimal communication
15. Use weights only for ranking and include qualitative information when this is necessary
16. Reproducibility
17. Allow for societal discourse on weights

This list is taken as the starting point here. Two requirements are added:

18. The weighting method should not contain any logical errors or contradictions.
19. The science used (including all types of science) should be the best available in the respective area.

Requirement no 5 can be operationalised by referring to the “check list” of impact categories presented in Table 1. Current practise for Life Cycle Inventory Analysis does however not include all these categories. Categories 2, 3, 5, 6 and 13 are typically not included at all or only to a limited extent (Finnveden, 1998). A minimum requirement for weighting methods is that they cover the impacts which currently are covered on most LCAs, i.e. categories 1,4, 7-12.

Valuation methods can be based on many different types of sciences. It is of course difficult, if not impossible to perform a comprehensive review of all different scientific aspects in all valuation methods, especially within a limited time and budget. In this paper, the focus will be on some aspects, whereas others will be largely uncommented. The attention given to the methods reviewed is not necessarily linked to the importance of the method or the amount of critics that could be raised. It is largely determined by the author’s ability to say something about the method which may be of interest for a wider audience. The EPS-system is given extra attention. The development of the EPS-system was financially supported by AFN, which is also supporting this study, and this is one reason for the extra attention.

2 Quantitative, operational weighting methods

2.1 Introduction

Based on Hofstetter (1996), Lindeijer classified existing weighting methods and approaches into 5 main groups:

- Proxy
- Technology
- Monetisation
- Authorised targets/standards (Distance-to-target)
- Panels

These five main groups will be discussed below.

2.2 Proxy approaches

Proxy approaches use one or a few quantitative measures, stated to be indicative for the total environmental impact (Lindeijer, 1996). Examples include energy requirements or total mass displacement (MIPS=material intensity per unit service). Clearly it is problematic to what extent environmental problems such as ecotoxicity and ozone depletion can be adequately covered by these

indicators (Lindeijer, 1996). Instead of attempting a weighting between different types of environmental problems, the proxy approaches pick one or a few. No inter-effect weighting is therefore included (this is one of the requirements listed in Table 2). This is an inherent limitation of the proxy approaches. Because no inter-effect weighting is included, the proxy approaches can not be described as a weighting/valuation method, and they will not be further discussed here. It can however be noted that some of these measures can be used as characterisation methods, e.g.
energy requirements or material used (e.g. Lindfors et al, 1995).

2.3 Technology

Technology abatement approaches are in most cases combined with some other measure, for example costs to reduce the burdens. In general, technology approaches are therefore included in other approaches. One exemption which may be described as a Technology method may be the Ecological Footprint.

The Ecological Footprint (EF) is a method for estimating the biologically productive area necessary to support current consumption patterns (Holmberg et al, 1999). It has generally been used for analysing nations and regions but can also be used as weighting method for LCA (Wackernagel and Rees, 1996). In contrast to other LCA-methods it has a starting point the area used by a system. This implies that it will capture other aspects than most other LCA weighting methods. Emissions are transformed into area by calculating the area needed for assimilation of the emission. This calculation is then dependent on the technology used for the assimilation and this is the reason why it is treated as a technology method here. In it's current versions, the ecological footprint can only handle very few pollutants, among them CO_2 and nitrogen emissions (Holmberg et al, 1999, Moberg, 1999). In its present version, there are thus large datagaps. It has also been argued that some aspects of human activities should not be included in an EF, for example emissions of persistent compounds foreign to nature (Holmberg et al, 1999). If so, some of the present limitations are inherent.

2.4 Monetisation methods

2.4.1 Introduction

There are a large number of different approaches for monetising environmental impacts. There is also a large number of ways of classifying the different approaches, sometimes leading to a somewhat confused discussion. The classification here is based on (but not identical to) other sources, (e.g. Turner et al (1994), Kågesson (1993) and Tellus Institute (1992)).

A first distinction can be made between methods that are based on "willingness-to-pay", and methods that are not. (Here is no distinction made between "willingness-to-pay" and "willingness-to-accept").

i. Methods based on willingness-to-pay.

The willingness to pay is normally related to the avoidance of something. Thus, somebody is willing to pay a certain amount of money in order to avoid something. This something can be early or late in the environmental mechanism. If it is late, the willingness to pay is to avoid a damage, if it is early, the willingness to pay is to avoid an intervention or a threat.

Environmental economists often distinguish between different types of values relating to natural environments. Again, the terminology is not completely agreed upon, the following is however based on Turner et al (1994). The first distinction is between use values and non-use values. The use values include both direct and indirect use values. An example of a direct use value is the timber value of a forest. The indirect use value includes the recreation value of the forest, the value of carbon fixation etc. (ibid.). Non-use values are non-instrumental values that are attributed to objects without the direct intention of actually using them (ibid.). Such values include the concern for, sympathy with, and respect for the rights and welfare of non-human beings. We may also attach a value to knowing that future generations will be able to enjoy use and non-use values (ibid.). Also the option to use an object, even if there is no intention of using it at the moment, may be attributed a value. In summary, the non-use value include option use value, bequest value and existence value (ibid.). The total economic value is the sum of the use and non-use values (ibid.).

A number of different methods may be used to derive a willingness-to-pay measure. Here a distinction will be made between three different approaches for deriving a willingness-to-pay-measure: 1) Individual's revealed preferences, 2) Individual's expressed preferences, 3) Society's willingness-to-pay.

i.i. Individual's revealed preferences

Methods based on individuals revealed preferences are assuming that people reveal their preferences in market prices. These methods thus assume that useful information can be derived from the market. It is usually damages that are valued in the market rather than interventions or threats. Thus, these methods usually need damage estimations. The revealed preferences are normally only related to the use values, and sometimes only the direct use value. Direct use values can often be derived from actual market prices, e.g. the market price of timber. Also the indirect use values may be derived from market values, though often indirectly. Examples of methods for evaluating total use values are the travel cost method and hedonic pricing methods. The travel cost method is a revealed preference method that can be used to estimate demand curves for recreation sites and thereby value those sites. These values are then

derived from people's travel costs. Hedonic pricing methods also attempts to evaluate environmental services by studying their influence on certain market prices. One example is house prices that are determined by a number of factors, including environmental aspects. Another example is wages that may vary depending on the risks associated with different types of jobs.

i.ii. Individual's expressed preferences

Non-use values can normally not be derived from revealed preferences (Turner et al, 1994). The contingent valuation method (CVM) bypasses the need to refer to market prices by asking individuals explicitly to place values upon environmental assets (ibid.). Because of this, the CVM is often referred to as an expressed preference method (ibid.). There are of course some similarities between CV-methods and the panel methods discussed below. In both, the answers will depend on who is asked. CVM may be used to value both interventions, threats and damages. In practise, it is most often used to value damages. This is because it may be easier to value something like "a swimmable river" (i.e. damage-level) rather than "emission of 1 kg of pesticide X" (i.e. intervention-level) (see also Söderqvist, 1995).

i.iii. Society's willingness-to-pay

A society's willingness to pay may be derived from political and governmental decisions. In these methods it is assumed that meaningful information on environmental values can be derived from political and governmental decisions.

One way of deriving a "societal price" is to study societies efforts to avoid a damage. An example may be the efforts to save a statistical life.

Another example of a method to derive a society's willingness-to-pay is to study the costs of reducing emissions to a decided emission limit. The marginal cost for removing the pollutant to the emission limit can be seen as the monetary value the society puts on the pollutant.

Yet another way of deriving a "societal price" is to look at "green taxes". If there are any taxes on emissions, these taxes may be seen as the societies willingness-to-pay (or rather willingness-to-accept) for that specific pollutant. Taxes are normally put on interventions (emissions and resource uses) rather than on damages. In some cases, charges are put on the steps before the emissions, that is the use of a product or chemical. Taxes are also used to raise money for the public budget. The fiscal motives behind the taxes does however not affect the valuation, because it is still a matter of priority to put a tax on something and to decide the size of the tax (Johansson, 1999).

ii. Methods not based on willingness-to-pay

There are also a number of monetisation methods that are not based on willingness-to-pay. They are often based on an estimation of a cost to do something, however if it is not clear that somebody is willing to pay this cost, it is not a measure of a willingness-to-pay.

A first example of such a method is a further development of one of the approaches mentioned above for evaluating a society's willingness to pay. In this approach the marginal cost for removing the pollutant to an emission limit is calculated. If the emission limit is a future target value, e.g. a critical load value (if such are available), it is no longer a willingness to pay that is evaluated (since it is not clear whether somebody is actually willing to pay), but another type of cost. Another example may be the cost for remidiation of a damage. This approach is of course only useful if remidiation is possible at all.

Since different monetisation methods cover different types of values, different methods should result in different results, and they do. For example, as a rule of thumb, the total economic value, as measured by the contingent valuation method, is typically an order of magnitude larger than the economic value derived from market valuations (KI, 1998). Therefore, a sum expressed in monetary units may not be directly comparable to another sum, expressed in the same units, because they may describe different values. A comparison can be made to the analysis of carbon. For wastewater, such analysis can be made on Total Organic Carbon, Total Inorganic Carbon, Dissolved Organic Carbon etc. For air pollutants there are also a number of measures. All these measures can be expressed as for example, mg Carbon. However, it is clear that they describe different things. Just because something is measured as mg Carbon does not make it immediately comparable to other measures expressed in the same unit. The same is true for monetary measures. Just because something is expressed in monetary terms does not make it immediately comparable or additive to another measure in the same unit. If a monetisation valuation method is used, the same method should therefore ideally be used to derive all economic values within the method.

In monetised valuation methods, a discount of future impacts is often made. A discount rate is often described as consisting of two parts (e.g. Turner et al, 1994). One is the pure time preference of the present generation. If impacts in the future are valued less important than impacts occurring today, simply because they are occurring in the future, the time preference is larger than zero. A time preference of zero is consistent with a view that future impacts are as important as today's. The other part of the discount rate consists of a function of growth of real consumption and the elasticity of the marginal utility of consumption (Turner et al, 1994). If we get richer in the future, the same cost will not be as important as it is today. Often the growth of real consumption is assumed to be exponential. However, if the capacity of growth is limited, a logistic growth may be more realistic resulting in a dynamic and very

different discount rate (Sterner, 1994). A positive discount rate, based on an expected growth rate, may be consistent with a view that future people are as important as current people are.

An explicit discount is not necessary for some types of economic valuations. For example, if the economic valuation is derived from a society's willingness-to-pay, an implicit discount has already been done. But if damage costs are to be estimated, a choice has to be made concerning future costs. The situation becomes acute if some damages are irreversible. In order to avoid infinite costs, either a discount must be used, or a cut-off after a certain time. In the latter case, all impacts occurring after the cut-off are neglected.

In the LCA-world, there is currently a couple of valuation methods suggested which are based on monetisation. These will be briefly described and reviewed below.

2.4.2 The EPS-system

The EPS-system has been described in a number of publications. This review is mainly based on Steen (1996) with other sources when relevant. Since the system is under development, later versions may include changes. The EPS-system is originally a one-step impact assessment method. According to Bengt Steen (personal communication, 1999), the EPS-system is not a weighting method. However, since it is often used as such, it will be included here.

The EPS-system is based on a valuation of "Safeguard subjects" (Steen and Ryding, 1992). These may be interpreted as "Areas for protection". Thus in reconstruction, the EPS-system may be seen as a two-step procedure in which a classification and characterisation is performed related to "Areas for protection", followed by a valuation of the results from the characterisation. The classification/characterisation step should then be based on natural science based information. The "safeguard subject classification list" in the EPS-system would then be (based on Steen and Ryding, 1992):

1. Biodiversity
2. Production
3. Human health
    a. Mortality
    b. Painful morbidity
    c. Other morbidity
    d. Severe nuisance
    e. Moderate nuisance
4. Resources

a. Minerals
b. Fossil fuels
c. Non-renewable fresh water
d. Buildings and installations
e. Art

5. Aesthetic values

In the EPS-system, the damage on these safeguard subjects from an emission should be quantified, and these damages are in a later step valued. For example, an emission of CO_2 will cause impacts on biodiversity, production, human health, resources and aesthetic values. All these damages should be quantified and in a later step valued in order to calculate the valuation weighting factor for CO_2.

The valuations are based on different types of measures (Steen and Ryding, 1992). The derivation of values of impacts on human health is not described in detail. The cited background data does however include both data from hedonic pricing methods and contingent valuation methods. Biodiversity is valued per capita as 10 times the amount the Swedish government is spending on conservation of biological diversity in Steen and Ryding (1992). No clear motivation of the number 10 is given. In the report by Steen (1996), the valuation is lower, still without a clear justification. Production losses are valued from market prices. Resources is valued based on assumed future costs (Steen, 1995).

In the EPS-system, different types of monetised measures are thus used for different problems. There is a mixture of market values (describing only a part of the total economic value) and other values covering larger part of the total economic value. This is a drawback from a theoretical perspective as discussed in section 2.4.1. The valuation of biological diversity is not documented. The changes between different in the valuation of biological diversity can be seen as a refinement. However, since no motivations are presented to the changes, it is difficult to understand the background for the refinement.

In the EPS-system (Steen and Ryding, 1992), no discount rates are used; instead the valuation is in some cases performed by integrating over a chosen time frame. For global warming, an integration over 100 years is used (Steen, 1996). For resources, the situation is however different. In this case, assumed future costs are used for the valuation (with current market prices) and no consideration is given to when these impacts could occur.

For many impacts, a cut-off is thus implicitly performed in the EPS-system. Impacts occurring after the chosen time-period are neglected. For example, impacts occurring from global warming after 100 years are not considered. The motivation for the time period is that the impact of an emission will last for about

100 years (Steen, 1996). This is not correct. A portion of an emission will remain airborne for thousands of years because transfer to the ultimate sink – ocean sediments – is very slow (IPCC, 1995). It takes approximately 1000 years to remove 85 % of the excess CO_2 from the atmosphere (Maier-Reimer and Hasselman, 1987). The impacts will continue for a longer time period. The global mean temperature is expected to continue rising for hundreds of years after a stabilisation of the concentration in the atmosphere, and after that, the sea level is expected to rise, at only a slowly declining rate for many centuries after the temperature have stabilised (IPCC, 1995). Impacts occurring later in the environmental mechanism as an effect of the rising temperature and sea level could continue for even longer time periods. For example, if climate change leads to large scale migrations of people, this is likely to give rise to political and social conflicts which might be troublesome even far into the future (Azar and Sterner, 1996). There is thus no evidence suggesting that the impacts will stop by the year 2100, or even that most of the effect will occur within the time period. In a study by Azar and Sterner (1996), the marginal cost of CO_2 more than doubled by increasing the time period from 300 years to 1000 years. The choice of a cut-off thus limits the impacts that are considered. Some chemicals contributing to climate change, have mean residence times of thousands of years in the atmosphere (IPCC, 1995). For these chemicals a cut-off after 100 years is clearly limiting the impacts considered.

The time period is only presented in the spreadsheets, not in the text describing the method, which is an illustration of the limited transparency. The use of cut-off is also inconsistent with the description of the EPS-system. It is stated that all generations have the same rights and should be given equal weights (e.g. Steen, 1997). In the operationalisation of the method, this is however not true since no consideration at all is given to impacts on future generations after the chosen time-period.

There are logical as well as computational errors in the calculations of different impacts and numbers having different units are added. Since the units of the different factors being multiplied are not always reported in a transparent way in e.g. Steen (1996), this is something which is difficult to evaluate. In personal communication Steen (1999) does however report the following units being multiplied for decrease of genetic capital as a result of CO_2-emissions:
$2*10^{11}$ [ELU/fraction of threatened species]* 2[fraction of threatened species]* 100 [years] * $1.3*10^{-16}$ [1/kg]
When these factors are multiplied the result is 0.052 [ELU *years/kg] and not 0.0038 [ELU/kg] as is reported by Steen (1996). This is one example of errors in the calculations. When numbers having different dimensions are added, the results are logically meaningless.

There are datagaps in the EPS-system. Some of these datagaps are apparent, whereas others require a detailed study of the data in the spreadsheets in order to be found. An example of the apparent datagap

concerns impacts from organic chemicals. It was concluded in Boström and Steen (1994) that the information available on chlorinated dioxins was not sufficient to calculate weighting factors. Since chlorinated dioxins probably is one of the best-studied groups of organic chemical, this means that it will be virtually impossible to include any hazardous organic chemicals in the EPS-system. An example of nonapparent datagaps is the calculation of the weighting factor for cadmium (Steen, 1996). Cadmium may cause human toxicological as well as ecotoxicological impacts. In relation to the latter it is generally recognised as an environmentally hazardous substance, e.g. in different lists from the Swedish National Chemicals Inspectorate (KemI, 1989). Despite this, no ecotoxicological impacts are considered. These datagaps are however not apparent. In order to identify them, a thorough review of the background material, in combination with knowledge on what should be included in a complete assessment, is needed. Thus only people with relevant expertise can evaluate the reliability of the damage assessments in the EPS-system. Such evaluations have apparently never been performed. This is a drawback of the EPS-system. It also illustrates the limitations of the transparency of the method. Datagaps are of importance since the uncertainty introduced by them can not be revealed in a sensitivity assessment.

The datagaps discussed above also illustrate some additional inconsistencies between the description of the system and its actual operationalisation. For example in Steen (1997) it is stated that the precautionary principle is a part of the system. It is however difficult to understand how the precautionary principle can be combined with the choice to exclude impacts because there is not enough knowledge about them. For example, according to Bengt Steen (personal communication, 1999) the chlorinated dioxins were excluded because no effects had been documented at that time. However, at that time the chlorinated dioxins had been recognised as hazardous substances for a long time (e.g. KemI, 1989, Bernes, undated). It thus seems like the demand for documentation of an effect to be included in the EPS-system is so high that a number of known hazardous substances can not be included. Another example of inconsistencies between the description and the operationalisation concerns the valuation of resources. They are valued based on assumed future costs. However, in the description it is stated that only actual impacts should be included. This seems contradictory. It is also difficult to understand how the statement that only actual impact should be included can be combined with the precautionary principle.

In the reports on the EPS-system, e.g. the latest publication with default values (Steen, 1996), data are presented without adequate references. This makes standard scientific reviews very difficult.

The EPS-system gives results that are not compatible with the common perceptions of the society. Braunschweig et al (1994) compared different LCA valuation methods. They applied the methods to an inventory with data corresponding to environmental interventions for the whole earth. They note that the

data are uncertain, and that the results should be interpreted carefully. Nevertheless, it is an interesting idea to compare different valuation methods on "global data". One of the results is that according to the EPS-system, the largest environmental problem is depletion of silver resources. It accounts for 26 % of the total environmental impacts of the earth according to the EPS-system. It is alone more important than for example, emissions contributing to global warming. This valuation may of course be true, it is not possible to know since we do not have the methods for analysing what the most important environmental problems are, partly because we don't know which the "right values" are, if there are any at all. But we can compare the results with the common perceptions of society and it does not seem like the EPS-system describes the society's valuations. It should however be noted that Braunschweig et al (1994) used an older set of weighting factors (Steen and Ryding, 1992). The more recent set (Steen, 1996) is different, but resources remain having a large influence on the results (e.g. Magnussen et al, 1998).

There are a number of other studies available in which the damage costs of different pollutants have been calculated. Comparisons between different studies and the EPS-system are thus possible. One example are calculations of the costs of some health damages from particulates, SO_2, VOC and NO_x by Cifuentes and Lave (1993). When these are compared with the valuation of the corresponding health effect in the EPS-system, the latter is lower by several orders of magnitude. A comparison can also be made with the calculations of damages to human health from respiratory effects made by Hofstetter (1998). Again the results from the EPS-system are several orders of magnitude lower. According to Hofstetter this can be explained by the missing impact paths and endpoints in Steen's study and the threshold behaviour assumed in the EPS-system. Via the study of Hofstetter (1998), the results of the EPS-system can also be compared to the results of the ExternE-method (IER, ETSU, EdM, 1997) (the ExternE project is briefly described in section 2.4.4). Again the results from the EPS-system are generally several orders of magnitude lower than the results from the ExternE-method. These differences suggest that there is a systematic difference in the calculations. Hofstetter has pointed out some reasons for the differences. A comprehensive review of the system by relevant specialists could further elucidate the reasons. No such review of the calculations and the data in the EPS-system has apparently taken place.

The use of the EPS-system can not be recommended today. This is for several reasons e.g. the mixture of different valuation methods, the limited transparency, the inconsistencies in the description of the system and the operationalisation, the datagaps, the lack of review of data and calculations, the lack of standard scientific reports with adequate references, errors in the scientific material, logical and computational errors, and the differences between the results and the results from reviewed publications. It should be noted that future developments of the system may of course change the conclusions drawn here.

2.4.3 The Tellus system

The Tellus system (Tellus Institute, 1992 with update, Zuckerman and Ackerman, 1994) uses data on society's willingness-to-pay to calculate valuation weighting factors. They use both data on emission taxes (the Swedish CO_2-tax) and marginal costs for reducing emissions down to decided emission limits.

The method contains valuation weighting factors for a limited number of pollutants to air (CO_2, CH_4, NO_x, SO_2, CO and Pb) plus a larger number of data of human toxicological relevance. For the latter group a specific characterisation method were used with the data for Pb as the equivalency factor. This characterisation method, in parallel to others for human health impacts, is discussed (see e.g. Lindfors et al, 1995, Jolliet, 1996, Finnveden and Lindfors, 1997, Udo de Haes et al, 1999 for recent reviews). The valuation weighting factors are however in principle open for recalculation using other human health characterisation methods. The limited number of pollutants included means that there are large datagaps for the method as a general method for weighting in LCA.

The Tellus method and data can not be recommended for use today. This is mainly because of large datagaps and the need for updating of valuation weighting factors and characterisation method.

2.4.4 Economic Valuation Analysis (EVA)/Impact Pathway Analysis

In a recent report, van Beukering et al (1998) discuss Economic Valuation (EVA) in Life Cycle Assessment. The data is used by Dobson (1998a and b) to develop a combined methodology based on LCA and EVA.

The economic valuation is largely based on the Impact Pathway Analysis (IPA) approach developed within the ExternE project (EC, 1995, Krewitt et al, 1998). There are some significant differences in the requirements for the IPA approach and LCA. In the IPA, the starting point is a known emission at a known site. The transports and chemical conversions are then modelled using site specific data to calculate the concentration and the deposition at specific receptors (humans, flora, materials and ecosystems). The damages to these receptors are then quantified. In the final step, the damages are valued and monetised.

In an LCA, the Impact Pathway Analysis will not be generally possible to use, since site specific information will not be available (see e.g. Udo de Haes, 1996), (a development towards a site-dependent assessment is however ongoing, see e.g. Potting et al, 1998). The data used by van Beukering et al (1998) was derived from specific sites. The applicability of these results to other sites is of course something that can be questioned. The idea behind the IPA approach is that there are significant differences between emissions at different sites. From this perspective it is not evident that the

results from one site can be used for a whole life cycle. Damage costs for some air pollutants emitted at different places in Europe can vary by a factor of 10 (Krewitt et al, 1998). It is not known to what extent the results from the specific site are representative for an average situation.

The IPA approach requires that the damage pathways can be modelled. So far, the IPA method has been applied to some air emissions (SO_2, NOx, PM10, VOC, As, Cd, Cr Ni and PAH (for the metals and PAH, only carcinogenic effects are considered). For waterborne emissions it has not yet been possible to use the IPA approach resulting in large datagaps. There are also datagaps with respect to the types of impacts considered. For example, for emissions of cadmium, only carcinogenic effects are considered. Other types of effects (e.g. kidney injuries (OECD, 1995)) are not considered. Furthermore, ecotoxicological effects are not considered at all.

In order to increase the number of pollutants considered, the characterisation method for carcinogens used by the Tellus institute were used (van Beukering et al, 1998). This choice of method could be updated.

The economic valuation is preferably done using data from contingent valuation studies (van Beukering et al, 1998). However, such data are not always available. In these cases other measures are used, e.g. costs that are incurred when impacts are treated (van Beukering et al, 1998). Such data give a lower bound of the real costs. In some cases, e.g. eutrophication, abatement costs were used (ibid.). As in the case of the EPS-system, the mixture of different valuation methods is a drawback from a theoretical standpoint (compare section 2.1).

For global warming, the IPA approach is not used. Instead data from the literature are used (van Beukering et al, 1998). The data is in the range between 5 and 25 euro/ton of carbon for impacts occurring until the year 2100. The problem of having a cut-off after one century was discussed above in connection with the EPS-system. There are other estimates in the literature which could be used and which are significantly different. For example, Azar and Sterner (1996) estimates the marginal costs at 260-590 US$/ton of carbon for a time horizon in the range 300-1000 years. This value is one or two orders of magnitude larger than the values cited by van Beukering et al. (For order of magnitude calculations it can be assumed that one euro equals one US$ equals 10 SEK). Major reasons behind this higher estimate is the choice of the discount model and a more accurate model of the carbon cycle (Azar and Sterner, 1996). The choice of literature data will thus have a major influence on the results and the choice should perhaps be further discussed and justified. When discussing monetisation based on damage costs of emissions contributing to global warming with damage costs, it is important to recognise that only effects which are possible to predict with reasonable accuracy can be included.

Surprises can be expected (IPCC, 1995) but are generally not included in the modelling. Despite the importance of analysing worst cases and surprises, they are in general not included in assessments made by IPCC and others (Johannesson, 1998).

The use of the EVA method as it is suggested by van Beukering et al (1998) and applied by Dobson (1998a and b) can not be recommended today. This is mainly because of datagaps, the choice of some of the costs (e.g. for global warming), and the choice of characterisation method for health effects.

2.4.5 Other approaches

The DESC-method (Krozer, 1992) are using costs of emission reductions to a certain target level as valuation weighting factors. The target levels may for example be "critical load levels" or targets of current environmental policy. There does however not seem to be any details, including weighting factors, about the method published.

There are some papers in which economists have applied an economic valuation to LCAs or LCAish studies (e.g. Craighill and Powell, 1995, Powell et al, 1996, Carlsson, 1997, and Sonesson et al, 1998). These are more ad-hoc studies in which economic values have been derived, mainly from literature, for the specific case study. It remains an open question to which extent these data may be used also in other case studies.

In a Dutch study (Huppes et al, 1997) “revealed preferences” were determined from government decisions by looking at costs deemed acceptable by governments for environmental problem reduction. This was done for six impact categories related to air emissions (greenhouse effect, ozone depletion, acidification, nutrification, summer smog, and human toxicity). This is an example of an estimation of a society’s willingness-to-pay. This was done in a comparison with the results from a panel study reviewed in section 2.6. Very few details are presented.

2.4.6 Non-reviewed approach

While this is written a new approach, Ecotax ’98, is being developed (Johansson, 1999, Johansson et al, 1999). In this approach the weighting factors are based on society’s willingness to pay, or rather willingness to accept, derived from different ecotaxes and fees in Sweden.

2.5 Authorised targets/standards (distance to target methods)

Several valuation methods are relating the valuation weighting factors to some sort of target (Lindeijer, 1996). These methods are conveniently called "distance-to-target-methods", although this name in some cases may be somewhat misleading. The major differences between different methods are

1) the precise shape of the equation relating the targets to the valuation weighting factors (see equations 1-4 below).

2) the choice of targets

3) whether inventory data or characterisation data, and if so, which type, are used in the weighting.

The simplest type of equation is

(1)

Where V is the valuation weighting factor and T is the target (expressed in units related to the target). Index i indicates either intervention i or impact category i. In the following, the indexes will be left out for simplicity.

In equation 1, the valuation weighting factor is inversely proportional to the target level. It is also implicitly assumed that all targets are equally important. It can be noted that this procedure is very similar to a normalisation. (It will in fact be a normalisation procedure if a wide definition of 'hormalisation" is used to include also "target levels" as "reference levels"). This equation was used by Schaltegger and Sturm (1991). The target levels were different types of quality standards, expressed as amount per mole of media (air and water). The valuation was performed on data from the inventory analysis.

Equation 4.1 may also be written as

(1')

where A is the actual flow within a specified area and time. The factor 1/A is thus a normalisation factor. Equation 1' was used by Baumann et al (1993) to calculate weighting factors for the "effect category method" (see also Baumann and Rydberg, 1994). The valuation is performed on results from a classification/characterisation method. The targets were either Swedish short-term political targets or long-term "critical levels".

Essentially the same equation (1') is used in the MET-points method (Kalisvaart and Remmerswaal, 1994). The targets are Dutch environmental policy goals for year 2010. The valuation is performed on results from a classification/characterisation method.

A development of equation 1 is to include a subjective weighting factor, W:

(2)

Equation 2 is identical to 1 if the subjective weighting factor is put to 1 for all interventions. Equation 2 was used by Corten et al (1994), putting all W to 1. In this study, the targets were related to impact categories instead of specific pollutants. The valuation was thus performed on results from a classification/characterisation method.

Equation 2 was also used by Goedkoop (1995) in the Eco-indicator 95 method with W set explicitly to 1. In this case however, the targets were related to environmental damages. The method is based on a two-step classification/characterisation method. The first step is a “normal” classification/characterisation, to a large extent based on the Dutch guidelines (Heijungs et al, 1992) but also including further developments. The second step is then, what can be called an "area for protection" classification/characterisation, i.e. estimations of damages are done. The following types of damages were considered:

* One extra death per million inhabitants per year
* Health complaints as a result of smog periods
* Five percent ecosystem impairment (in the longer term)

These three "targets" were then considered equally important.

Thus the same equation can result in very different valuation methods depending on the type of information used.

Another equation is suggested in the Ecoscarcity method (Ahbe et al, 1990)

(3)

In this method both the target and the actual levels are related to a given region or country. The target levels are derived from annual load targets as set by national environmental protection agencies, laws and regulations. The valuation weighting factor, V, is to be multiplied with data on environmental interventions. No classification/characterisation is thus performed. The valuation weighting factors were originally calculated based on Swiss data. Since then, the method has however been adapted to a number of other countries (see e.g. Lindfors et al, 1995) and it has recently been updated (BUWAL, 1998).

Kortman et al (1994) discuss distance-to-target methods and suggest the equation

(4)

for A > T and V=0 for A < T. The factor 1/A is the normalisation factor. The targets are suggested to be the "No significant adverse effect level" (NSAEL). The method is based on results from an "environmental threat" classification/characterisation. Also in this method W was preliminary set to 1. A number of other equations can be envisaged. Some alternative ways of expressing the "ecoscarcity function" has for example been discussed (Ahbe et al, 1990).

In the distance-to-target methods, the targets are normally related to either political/administrative target levels or "critical" or "sustainable" levels. A first problem is then to define the targets.

In the case of political/administrative targets, there may be several types of targets. There are for example targets related to the environmental quality, targets related to environmental interventions and threats, and targets on flows inside the technical system. There may also be different targets related to different areas for protection, e.g. quality standards relating to drinking water may be different than standards relating to the water quality of oligotrophic lakes in national reserves. There may also be targets relating to different time frames: short term targets, long term targets and targets without any specified time frame. Targets may be decided by different authorities, the government, the parliament, in international conventions etc. Since different targets are decided by different groups and with different aims, they may not always be compatible with each other. When developing distance-to-target methods, a choice must thus be made concerning which types of targets to go for. There may be problems in finding targets for all types of relevant environmental problems which are compatible with each other.

To relate the targets to "critical" or "sustainable" levels is difficult and probably impossible. Some attempts have been made (e.g. Baumann et al, 1993, Kortman, 1994). However, when the data are scrutinised it seems like in the end different types of more or less arbitrary, administrative decisions are taken as the basis for the level. This is not surprising since although "critical levels" has been established in some areas, it will not be possible to calculate it for all types of environmental problems (Chadwick and Nilsson, 1993). In practise: critical loads are difficult to establish because (ibid.):
1. In many situations knowledge is too limited to allow quantitative limits to be set.
2. The no-effect level is zero or close to zero.
3. Problems of scale both in relation to dose and response.

In the Eco-indicator 95 project (Goedkoop, 1995) the "targets" are chosen in another way. They are explicitly defined in order to achieve an equivalency. The justification of the equivalency of the three "targets" is however largely lacking and the choices seem rather arbitrary. By the choice of the "targets", the equivalency between "impaired ecosystems" (in terms of area) and number of deaths will depend on the population density, which of course varies e.g. between the Netherlands and Sweden. It is also

somewhat unclear how "impairment" should be defined both in terms of the magnitude of the impact and in time. The Eco indicator 95 has been adapted to recent knowledge about environmental impacts by Frischknecht (1998). The Eco indicator is currently being updated as discussed below and a new version, Eco indicator 98, will be published (Goedkoop et al, 1998).

There are several versions of distance-to-target methods available including different equations for relating the target to the valuation weighting factor. Although arguments may be raised for and against different equations, there is (at least not for the moment) no way a rational choice between those discussed above, and others, can be made. This introduces a certain arbitrariness to the results.

The available distance-to-target methods are all based on the assumption that all targets are equally important. This is a critical assumption, which apparently has never been justified. If the targets are political/administrative targets there is no reason to assume that rational decision makers will decide target levels in such a way that are all equally important. This is so because this is normally not a requirement when decisions on target levels are made. In addition target levels should not only be based on what is environmentally important but there are also other considerations to be made, e.g. costs (Wenzel et al, 1997). This can result in situations where a target only to a limited extent reflects the importance of the impact, but to a larger extent the costs associated in reaching the target. If the target is a function of the importance and the costs, the importance will be a function of costs and targets. This suggests that if we are interested in the importance of the impacts, this should be estimated from a combination of information on targets and costs, and not targets alone. If a combination of targets and costs are used, the result will be one type of monetisation method based on a society's willingness-to-pay, discussed above in section 2.4.1.

If the targets are based on levels which are considered "critical" or "sustainable" there is again no specific reason why all targets should be equally important. Although the choice to assume that all targets are equally important may appear as a neutral and non-subjective choice, it is in principle no less subjective or arbitrary than any other weighting unless it can be argued that targets are set in such a way that they should be regarded as equally important (this case is further discussed below). By assuming that all targets are equally important, the distance-to-target methods are in a sense avoiding the explicit weighting. This is in contrast to the requirement in Table 2 concerning the inter-effect weighting. If all targets are assumed to be equally important, no weighting between the targets are done, and no inter-effect weighting is being made. The conclusion from the discussions on Life Cycle Impact Assessment within the LCANET (a Concerted Action in the Environment and Climate Programme) was that distance-to-target methods should not at all be regarded as valuation weighting methods (Finnveden and Lindfors, 1997). Instead it can be considered as a sort of normalisation procedure.

In the EDIP method a similar conclusion is made (Wenzel et al, 1997). Equation 1 is used with political targets for the year 2000 for pollutants. However, the results are never aggregated into a single score. Instead the results are presented using the same impact categories as was obtained after the characterisation.

If the targets are set in a procedure where they are explicitly set to be equally important, the inter-effect weighting is taken place in the target-setting, and the distance-to-target method can be described as a weighting method. The target setting must then be made in a formalised procedure, e.g. a panel approach, or by using some sort of monetary measures. The method can then be described as a combination of distance-to-target methods and panel methods or monetisation methods.

None of the distance-to-target methods can be recommended as a valuation weighting method. This is mainly because they are not valuation methods. This is because no inter-effect weighting is made. The Eco-indicator ´95 is an exception. Here the targets are explicitly assumed to be equally important. The justification for this assumption is however limited and because the method is updated anyway, it can not be recommended now.

2.6 Panel methods

Panel approaches are used as a heading for a number of different approaches which have one thing in common: people are asked to give weighting factors. Panel methods can be distinguished according to several criteria (Brunner, 1998):

- Method, a panel study can be done with questionnaires, interviews or group discussions.
- Group of panellists: Experts, stakeholders or laypeople.
- Procedure: The panel can be done in a one-round procedure or in a multi-round procedure with feedback (e.g. Delphi-method).
- Outcome: The goal can be a consensus or the answers are analysed by statistical methods.

In connection with LCA, panel methods have mostly been developed on an ad-hoc basis in connection with a specific case study. Examples of studies in which quantitative panel methods have been used are

Anonymous, 1991, Kortman et al, 1994, Huppes et al, 1997, Wilson and Jones, 1994, Nagata et al, 1995, Lindeijer, 1997, Poulamaa et al, 1996 and Goedkoop et al, 1998.

In a Dutch study (Anonymous, 1991, Annema, 1992) the starting point was normalised characterisation results. The application of weighting factors was then done in a Delphi-like process. In the weighting, members of the steering group representing industry, government, environmental groups and some independent persons from universities and scientific institutes were involved. The process was a four-step approach. The aim of the first step was to gain a common understanding of the importance of the impact categories and of facts that were included in the environmental profiles. One basis for the discussion was a framework in which different aspects of the different categories were defined such as whether the impact is only on humans or ecosystems or both, the degree of scientific uncertainty, the degree of reversibility of the impact, the scale of the impact, the timing of the impact and other issues. The second step of the process was a first assessment of the weighting factors. Each member confidentially did this step. In the third step, the results were presented to the members who continued the discussions. The fourth step was a second assessment. The process was then continued until a ranking had been produced.

In a second Dutch study (Kortman et al, 1994), 22 environmental experts were interviewed. They were given information about the environmental problems, then asked to rank them. The environmental problems included were: Greenhouse effect, acidification, ecotoxicity, ozone depletion, nutrification and human toxicity. In a second step they were asked to divide 100 points over the environmental effects in such a way that the distribution of points reflects the relative seriousness of each effect in the rank order. The results from an alternative valuation method based on a distance-to-target method were presented to the experts who were given an opportunity to reconsider their earlier answers.

In a British study (Wilson and Jones, 1994), the valuation was performed directly on the inventory data using a Delphi technique. A panel of eleven anonymous experts from British Universities was used. They gave their views on the subject being investigated by completing a questionnaire. The results of the survey were summarised and fed back to each expert by post showing how his or her view differed from the other participants. The experts were then invited to reconsider their positions. From the judgements obtained in the second iteration, scores reflecting the median values were obtained which were then applied to the inventory data.

In the third Dutch study (Huppes et al, 1997), the panel consisted of representatives from the ministries involved, specialists from companies affiliated to NOGEPA (Dutch Oil and Gas Exploration and Production Association), and a number of external specialists. Only air emissions were considered. They

were asked to pass judgement under Rawls' "veil of ignorance" (Rawls, 1972), and therefore not as representatives. The environmental problems were specified in space and time. After a session with elucidation of the relevant environmental problems by specialists, each participant determined his weighting factors, by dividing 100 importance points over the six impact categories. Following discussions and questions, three further weighting rounds were held. The results after the final round were then used in the case study aiming for prioritising investment options (Huppes et al, 1997).

A Finnish study aimed at developing an index for "Environmental Friendliness" which can be used in a number of different applications, among them LCA (Puolamaa et al, 1996). The weighting included the following categories (in order of importance according to the study): greenhouse effect, ozone depletion, loss of biodiversity, radioactivity, ecotoxicological effects, resources depletion, photo-oxidants, acidfication of soils, eutrophication, acidification of waterbodies and noise exposure. The study was done with interviews using the Analytical Hierarchical Process (AHP) in which the respondents are asked to make pairwise comparisons. It can be noted that the procedure does only allow answers in which the most serious problem is nine times more serious than the least serious problem.

In a Japanese study, Nagata et al (1995), a questionnaire was filled in by 242 panellists. Later Lindeijer (1997) sent an extended version of the same questionnaire to European panellists. The questionnaire covered different subjects (selection and weighting of safeguard subjects, weighting of impact categories etc) and different weighting methods (ranking methods, scores allocation method and AHP) (Brunner, 1998).

These examples illustrate different approaches for panel methods. Another technique used in decision theory which has been discussed in connection with LCA is the multi-attribute utility theory (MAUT) (e.g. Fava et al, 1993, Hofstetter, 1996, Miettinen and Hämäläinen, 1997).

One of the lessons from studies on panel methods is that the results are sensitive to how the questions are being asked (Heijungs, 1994, Kortman, 1994). Also apparently straightforward questions can be interpreted in different ways (ibid.). The difficulties in interpreting questions are also illustrated in a study by Landbank (Wilson and Jones, 1995) in which 17 experts were members of a panel. Answers from 7 of the experts had however to be disregarded since it was apparent from the answers that the respondents had not understood the question (ibid.). This is also illustrated in a study by Mettier (1999). Panellists were asked to weigh three safeguard subjects: Resources, Ecological Quality and Human Health. Panellists were also asked to weight hypothetical products. From information about the hypothetical products, the corresponding scores for the safeguard subjects could be calculated and compared to the weights given earlier. Interestingly, very different results were obtained. One reason for

this can be the tendency to assign more or less equal weights to all criteria to be aggregated (Hofstetter, 1999, based on Mettier (1999) and Lundie and Huppes (1999)), more or less regardless of what is to be weighted. It thus seems like the results are largely determined already when it is decided what is to be weighted against each other.

One reason for the tendency to give similar weights to all aspects to be weighted is methodological. In some studies the panellists are asked to distribute 100 points between different objects. In the AHP process used by Puolamaa et al (1996) the maximum difference possible is a factor 9. Since there is no reason a priori to assume such equal distribution, this seems to be a major drawback of these studies.

Cognitive stress is probably a frequent reaction to existing questionnaires and the results derived from them have to be interpreted carefully (Hofstetter, 1996). According to Brunner (1998) cognitive stress is a major problem with the studies performed by Wilson and Jones (1994), Nagata et al (1995) and Lindeijer (1997). According to Hofstetter (1996) other problematic points are: the way the weighting problem is presented to the panellists, the differences in information that individuals possess on environmental mechanisms and problems, the different weighting criteria applied by each panellists, and the distortions that occur when averages are formed of individually independent weights or through consensus processes. Brunner (1998) notes that too little emphasis has typically been put on the selection and number of panellists. Another point made by Brunner is that the used models of environmental effects and impact categories were not always consistent

One of the studies in which panels have been used and one-step weighting factors have been published is described by Wilson and Jones (1994). The weighting factors derived by the panel are somewhat surprising. According to the expert panel, 1 kg of high-level radiation waste is approximately as bad as 1 kg of CO_2 and 6 kg of CO_2 is as bad as 1 kg of mercury emitted to water. It can be anticipated that many experts, as well as common people, would have different opinions. This valuation may of course be true, it is not possible to know since we don't know the right answer (or even if there exist something which can be called "the right answer", but it does not seem like the method describes the society's valuations.

None of the datasets so far presented from panel methods can be recommended for use. This is because they have in general been developed for specific cases and the applicability to generic use is not clear, there are significant datagaps in many of them, for some of the methods some of the results are quite strange, and the methods have often been designed to give more or less equal weights to all object to be weighted.

When this is written, the Eco-indicator 98 has not been published. In the method, a damage approach will be used where damages to resources, ecosystem health and human health are modelled based on natural science (Goedkoop et al, 1998). These damages are to be weighted against each other in a panel approach. The panel members are assigned to one of three value systems based on Cultural Theory (Mettier, 1999). (Cultural Theory is a model which characterise societal groups and their world views which can be used to model value choices inherent in LCA in a way that is compatible with the values and beliefs of a number of archetypes (Hofstetter, 1998)). The result will therefore be three different indicators. The cultural theory has also been used for the damage assessments (Goedkoop et al, 1998). When subjective choices are encountered in the natural science based modelling, different choices are made, for the three different indicators, based on the respective value system.

3 Qualitative methods

The focus in this paper is on quantitative methods. Qualitative methods will be reviewed mainly as an information on published approaches, rather than as a critical review.

Qualitative valuation methods can often be seen as methods for structuring information provided by the LCA. The aim is often to facilitate for somebody to draw conclusions from the study. Different types of matrices will often be useful.

One group of matrices, for example used by Christiansen et al (1990) and Graedel et al (1995), consists of the life-cycle stages on one axis and environmental parameters on the other. The latter may be inventory parameters or some type of classification parameters. This type of matrix may be called a Life-Cycle matrix, Fig. 2.

Fig. 2 Example of a Life-Cycle Matrix.

	Global Warming	Human toxicological impacts	.....	.....
Raw material extraction				
Production				
.....				
.....				

The matrices has been used both for absolute and comparative analysis. In an absolute analysis, a sign or a number indicates the severeness of an impact. A major problem is to find criteria for the severeness. It can therefore be argued that a comparative analysis is to be preferred. In a comparative analysis, the impact of one alternative is compared to a reference case, indicating by signs or number whether the alternative scores better, worse, or equal with respect to a specific environmental problem for a specific life-cycle stage.

A similar matrix can be constructed based on the MECO-principle (Wenzel et al, 1997, Wenzel, 1998). Instead of impact categories, the following headings can then be used M (Materials), E (Energy), C (Chemicals), O (Others).

A major problem with a comparative analysis is the difficulties in comparing the relative importance of the elements in the matrix. It may be tempting to assume that all elements are equally important and to believe that this is a neutral decision. It is however an arbitrary and subjective decision, just as arbitrary and subjective as for example assuming that one of the elements are one thousand times as important as another one. Therefore these types of matrices may be difficult to use when the aim is to make an overall evaluation, although they may be very useful for illustrating a decision situation.

In the "Verbal-argumentative approach", used by Schmitz et al (1994) (also described by Giegrich and Schmitz, 1996), the relative importance of the different environmental impacts are analysed in a separate and more structured way. When determining the "ecological importance" of the different impact categories, five different criteria were used:

* Ecological threat potential
* Reversibility - Irreversibility
* Global, Regional, Local
* Environmental Preferences of the Population
* Relationship of Actual and/or Previous Pollution to Quality Goals

Based on these criteria, the "ecological importance" of the relevant impact categories were evaluated verbally in five categories ranging from "lesser importance" to "very large importance".

When working with a qualitative approach, other criteria may of course also be chosen. Again, a matrix may be useful for structuring the discussion. In this second type of matrix, the environmental problems are on one axis and the chosen criteria are on the other axis. An example of a matrix for determining "environmental importance" is presented in Fig 3.

Fig 3. Example of a matrix for determining the environmental importance of an impact category.

	Global warming	Human toxicological impacts	.....	.....
Reversibility				
Geographical scale				
.....				
.....				

In the method used by Schmitz et al (1994), the results from the characterisation were normalised and presented both quantitatively and verbally, the latter within five categories ranging from "small" to "very large". Based on the normalised results and the evaluation of the "ecological importance" of the different parameters, a verbal-argumentative overall evaluation was performed and conclusions were drawn.

Volkwein et al (1996) have suggested another semi-quantitative approach. The method consists of four steps. In the first a quantitative normalisation is performed on the difference between the two alternatives compared. In the second step three qualitative criteria are used: space, time and hazard for different impact categories. In the third step a formulised ranking procedure is performed based on the three qualitative criteria and the results from the normalisation. In the last step a final judgement is made. The last four steps are suggested to be performed by an expert panel.

The results from a qualitative valuation will often not be reproducible since other persons performing the valuation could possibly reach another conclusion. This means that the choice of persons to perform the valuation will be important for the overall result.

4 Conclusions and Discussion

Although there are several quantitative, generic valuation weighting methods available, none can at this stage be recommended for use today. This is mainly because they either suffer from significant datagaps, include inconsistencies, lack justification of important assumptions, or in the case of most distance-to-target methods because they do not include an inter-effect weighting and are therefore not valuation methods at all although they often are used as such.

It can be noted that the conclusion above is not dependent on the application of the LCA. It can also be noted that the conclusion is not based on an evaluation of the values behind the different valuation methods. Furthermore, the conclusion is not based on an evaluation of the need for valuation methods (the latter is discussed e.g. in Finnveden, 1996). It is taken as a given thing that some people in some applications want to use valuation methods.

Since there is no method that can be recommended today, there is an obvious need for further research.

A major limitation of the distance-to-target approaches is that they do not in general include an inter-effect weighting. The exception is when the targets are explicitly set to be equally important. However, this target setting must be done in a formalised procedure, e.g. in a panel approach. The distance-to-target methods which do include an inter-effect weighting are thus used in combination with some other approach. The “pure” distance-to-target methods will however not include an inter-effect weighting. This limitation seems to be inherent suggesting that it will not be possible to develop weighting methods within this approach which fulfils the requirements. Further research concerning “pure” distance-to-target methods may therefore prove to be improductive and a dead-end.

It is suggested that further research is mainly directed towards monetisation and panel approaches. Many of the limitations of the presently available methods within these groups do not have to be inherent and it should be possible to minimise them through further work. Other approaches could be used in combination. Ideally, research should be made in co-operation between LCA specialists and other specialists from relevant fields. It is clear that the LCA community can learn from other areas, e.g. economy and decision theory (e.g. Miettinen and Hämäläinen, 1997). However, LCA puts some special requirements on the methods used, and the necessary adoption will probably need knowledge also of LCA.

Most recent work on valuation methods has focused on damage approaches (e.g. ECO-indicator 98, and economic valuation based on the Impact Pathway Analysis). This work should be continued. There

are however some inherent problems with damage approaches. They require that the damages can be modelled. This is very difficult, or even impossible, in some cases. If only known impacts are considered, a damage approach will lead to underestimations of the environmental impacts. In a damage approach there is little room for the precautionary principle. A suggestion has however been made by Hofstetter (1998) that the model of the ecosphere should not only include the known damages, but also a proxy for the unknown damages, and also an indicator of the manageability of the damage. The introduction of an indicator for the unknown damages is commendable since it makes the problem explicit. There is of course a problem of finding a good proxy for the unknown damage. The suggestion by Hofstetter is to use the bioconcentration factor in fish (BCF) normalised with the global anthropogenic emissions and geogenic flows. The BCF is a well-established indicator for ecotoxicological impacts. It has also been suggested as a characterisation parameter for ecotoxicological impacts (Finnveden et al, 1992). However, it remains unclear to what extent it is a useful proxy also for other types of unknown damages. It will for example probably not be useful in describing surprises and unknown effects resulting from global warming briefly discussed in section 2.4.4.

Another inherent problem of the damage approaches is the question how to handle future impacts. The choices concerning discounting and/or a cut-off can have a decisive influence on the results.

Due to the problems of damage approaches, research on valuation methods should also include approaches based on valuations on an intermediate level in the environmental mechanism, i.e. on the level of environmental threats rather than actual damages. Such approaches could include both economic and panel methods. Economic methods, which could be further developed, include different approaches for estimating the society's willingness to pay.

Another important question which must be raised is whose values should be used when deriving weighting factors, and when choosing the method for deriving weighting factors. Different groups and stakeholders are likely to have different values and worldviews, not only influencing the weighting factors, but also which method they will find acceptable. If robust results are wanted, it can be important to use several methods and sets of weighting factors to examine the sensitivity of the results to different values and worldviews. Discussions on how values and worldviews can influence methodological choices have been published in several recent theses (Finnveden, 1998, Hofstetter, 1998 and Tukker, 1998).

Since no valuation method can be recommended today, it is still an open question what to do in a specific LCA case study. First of all, the need for a weighting method should be considered. In many applications, the need for a weighting method is limited in order to be able to draw conclusions

(Finnveden, 1996). Another recommendation sometimes made is that if valuation methods are to be used, several should be used in parallel (e.g. Lindfors et al, 1995a). This is probably a prudent choice. However, one of the major problems with the present methods is that there are significant datagaps. Since many methods have datagaps in the same areas, e.g. concerning emissions of toxicological relevance, the results should be cautiously interpreted even if all applied valuation methods point in the same direction. If the valuation methods give different results, as they often will do, the question still remains concerning how to draw conclusions. The added information of performing a valuation may in these cases be quite low.

When choosing methods to use in combination, one could try to find methods which are complementary regarding methodological approach and coverage of impact categories. One should also look for methods which at least fulfil some basic requirements, e.g. no logical errors and at least some inter-effect weighting. Among the presently available monetisation methods, an economic evaluation based on the ExternE project is probably the preferable choice. In comparison to the EPS-system it does not include any apparent errors, and in comparison with the Tellus system it is more updated and comprehensive. Among the distance-to-target method, the Eco-indicator ´95 is the only method which includes an inter-effect weighting (this was done when choosing the targets since they were explicitly chosen to be equally important), which probably makes it the most reasonable choice. Among the presently available panel methods, the study by Puolamaa et al (1996) is probably the study that followed the rules of panel studies most accurately according to Brunner (1998), and is therefore probably the best choice. To my knowledge there is no study in which these three methods have been used and compared. An economic valuation based on the ExternE methodology have however been used in combination with the Eco-indicator ´95, e.g. by De Nocker et al (1998) in a study on fuels. The comparison illustrates that the results differ a lot, but can be regarded as complementary. When choosing methods, newly developed methods, such as the Eco-indicator ´98 and Ecotax ´98 could also be considered.

Weighting methods in LCA remains a controversial issue. As noted in section 1.4, it is difficult to critically review valuation methods. It is sometimes easier and more acceptable to be critical about the valuation element as such. From personal experience I know that this has led some people to direct their criticism towards the valuation element, when they in fact were critical about a specific valuation method. However, I believe that we should critically evaluate also weighting methods. There is no reason to accept bad science or errors, just because it is a part of a valuation method. I believe it is possible to develop much better weighting methods than the ones that are available and used today. While waiting for better methods, it is important not to be paralysed. As discussed elsewhere (Finnveden, 1998), the uncertainties involved in the valuation is only one reason why it in fact will be difficult to show that one product system is more or less environmentally friendly than another one, even in situations where this is

actually the case. This has policy implications. If we want changes to be made, we must learn to take decisions and act on a less rigid basis than full scientific proofs.

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