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RAPPORTSAMMANFATTNING
Report Summary

Projekttitel/Project title

Anslagsgivare för projektet/Project sponsor

Rapportförfattare, author
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Rapportens titel och undertitel/Title and subtitle of the report
Valuation methods within the framework of Life Cycle Assessment

Sammanfattning/Summary

Life Cycle Assessment Valuation methods are discussed. Different approaches for valuation are discussed as well as presently available valuation methods in relation to:

* the values involved in the valuation
* the LCA framework
* different applications of LCA.

Among the conclusions are:

* ethical and ideological valuations are involved not only when applying valuation weighting factors, but also when choosing valuation method and also when choosing whether to perform a valuation weighting or not
* it can be questioned whether straight distance-to-target methods are valuation methods
* it is still an open question whether presently available valuation methods produce meaningful and reliable information
* further development of quantitative valuation methods could concentrate both on different types of monetarisation methods and panel methods
* in many applications of LCA, the expected result is an identification of critical areas rather than a one-dimensional score, reducing the need for valuation methods

Nyckelord samt ev. anknytning till geografiskt område, näringsgren eller vattendrag/Keywords
LCA, Life Cycle Assessment, Impact Assessment, Valuation, Methods, Review

Bibliografiska uppgifter/Bibliographic data
IVL Rapport B1231

Beställningsadress för rapporten/Ordering address
IVL, Publikationsservice, Box 21060, S-100 31 Stockholm, Sweden

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1 Introduction

1.1 The framework

A Life Cycle Assessment (LCA) is a process to evaluate the environmental burdens associated with a product, process, or activity by identifying and quantifying energy and materials used and wastes released to the environment; and to assess the impacts of those energy and material uses and releases to the environment. The assessment should include the entire life-cycle of the product, process or activity encompassing materials and energy acquisition, manufacturing, use and waste management (modified from Consoli et al, 1993).

During the last years there has been large developments in LCA methodology and a common framework and terminology has emerged. A 'Code of Practise' (Consoli et al, 1993) and several guidelines (e.g. Heijungs et al, 1992, Vigon et al, 1993, Lindfors et al, 1995b) have been published and an ISO standard is currently being developed.

An LCA may be divided into three components and the third into three (or four) subcomponents:

* Goal Definition and Scoping, where the goal and scope of the study are defined.
* Inventory analysis, where the in- and outputs to and from the system under study is compiled.
* Impact assessment, divided into
    - classification
    - characterisation
    (- normalisation)
    - valuation

The classification subcomponent is a qualitative step in which the different inputs and outputs are assigned to different impact categories based on the expected types of impact on the environment. The classification should be based on a scientific analysis of the relevant environmental processes (rather than administrative criteria). The classification should thus answer the question: Which are the expected environmental impacts of each input and output of the system?

The characterisation subcomponent is mainly a quantitative step in which the relative contributions of each input and output to its assigned impact categories are assessed, and the contributions are aggregated within the impact categories. The characterisation should be based on a scientific analysis of the relevant environmental processes. The characterisation should thus answer the questions: What is the potential contribution of a specific input or output to different environmental impacts? and What is the total potential contribution of the system to different environmental impacts?

Sometimes the performance of a normalisation step is suggested, in which the data from the characterisation are related to the total magnitude of the given impact category in some given area and time (Consoli et al, 1993). The normalisation may be performed as a separate subcomponent, or as a part of either the characterisation or the valuation.

The valuation sub component is either a quantitative or qualitative step in which the relative importance of the different potential environmental impacts from the system are weighted against each other. The valuation will involve political, ideological and ethical values.

The valuation sub component may be interpreted in two different ways, either as the subcomponent in which valuation methods are used, or as the subcomponent in which conclusions are drawn. Here, the first interpretation will be used. The distinction is useful since it suggests that it may sometimes be possible to draw conclusions without the use of valuation methods and also that the conclusions drawn from a complete LCA may be different than the ones suggested by the valuation methods. In order to further stress the distinction, valuation methods will here be interpreted as methods resulting in a one-dimensional score (in the case of quantitative methods) or a single judgement on the environmental preferences of one system over another.

The framework outlined above will be further discussed in chapter 3.

1.2 The present study

Several methods for valuation in connection with LCA has been developed during the last 5 years and are being used. Several reviews and discussion papers on valuation methods have been published recently (e.g. Lindfors et al, 1995a and b, Lindeijer, 1996, Giegrich et al, 1995, Braunscheweig et al, 1994). 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).

This paper will discuss different approaches for valuation methods. It will start in chapter 2 with a brief discussion about the ideological and ethical values that may be involved in the valuation. In chapter 3, the framework for LCA as outlined above will be discussed in relation to the question: What type of information is valued (i.e. information from the inventory analysis, characterisation, etc.)? In chapter 4, different approaches for actually performing the valuation will be presented. Presently available valuation methods are critically discussed in chapter 5. LCA may be used for different applications. In chapter 6, some of these will be discussed in relation to the need for different types of valuation methods for different types of applications. Chapter 7 includes a final discussion and conclusions.

2 Values in the valuation

2.1 Introduction

It is often stated, as for example above, that the valuation in LCA must include political, ideological and ethical values. However, these values are rarely discussed. An "ideology" consists of at least two aspects: a perception of how the world is and an ethical standpoint, i.e. ideas on how the world should be (e.g. Hansson, 1981). Here, an attempt will be made to briefly discuss some ideological standpoints that may influence the valuation in LCA. It is suggested that this influence can be on three levels:

1. Should a weighting be performed at all or should some aspects be given absolute priority?
2. If a weighting is to be performed, which methodological approach should be chosen?
3. Given a certain methodological approach for weighting, which are the valuation weighting factors?

A zeroth level can also be distinguished. This is the level concerning if an LCA is to be performed at all. By performing an LCA, it is implicitly assumed not only that the information it will produce is somehow useful but also that it should be used. This level will however not be further discussed in this report.

It should be noted that the discussion here will not be on ideological differences in views on how important environmental issues are compared to other issues in society. In the valuation in LCA, this is not of prime interest. Of concern is instead how different environmental aspects are valued against each other.

2.2 Weighting methods or not

Ethical theories can broadly be divided into two groups: deontological and teological (e.g. Frankena, 1963). According to the teological theories, the ethical value of an act is related to the consequences of the act. The ethically right acts are those that produces the best consequences. According to utilitarianism (which is a teological theory), it is the universal consequences that should be considered. Different schools have different opinions on what the "best" consequences are. For example, hedonistic utilitarianists consider consequences leading to the maximisation of pleasure over pain to be the ethically good.

The deontological theories are not primarily related to the consequences of the acts. Other criteria must therefore be used. Examples of such criteria are authoritative rules such as God's 10 demands or United Nation's declaration of human rights (Ariansen, 1993). According to other deontological theories, the ethics are related to the procedure by which the normative rules are developed. If the procedure by which the rules are determined is ethically acceptable, than the rules will also be acceptable. One often cited example is "the theory of justice" developed by Rawls (1973).

One important difference between the teological and deontological theories is that according to the teological theories, trade offs are possible, i.e. a weighting between different good and bad consequences is possible. According to the deontological group of theories, there will in general be rules which are given absolute priorities and trade-offs are therefore not possible.

This has direct relevance for the discussion on valuation methods in LCA. For persons holding ethical beliefs in line with deontological theories, methods involving weighting of different environmental problems against each other may seem unethical. Valuation methods involving weighting are on the other hand in line with teological theories such as utilitarianism.

Persons holding deontological ethics may however still accept a utilitaristic approach for some type of decisions. For example, Prawitz (1990) argues that the maxim of maximising expected utility (which is in line with utilitarianism), is not acceptable when decisions are to be taken in which one alternative may lead to a negative outcome of catastrophic proportions. If this is not the case, expected utility may well be an acceptable decision criteria (*ibid.*). In relation to LCA, a reasonable standpoint for some people may thus be that as long as the LCA is concerned with small scale decisions involving only small changes, a weighting procedure may be acceptable. For larger decisions, involving larger consequences, it may not be OK.

There are only few valuation methods developed which does not involve a weighting procedure. However, Pedersen Weidema has developed two such methods for LCA. In the first (Pedersen, 1991) an absolute priority is given to irreversible use of non-renewable resources which also by themselves are non comparable. The second involves a further development of Rawl's 'Theory of justice' (Pedersen, 1992) in which an absolute priority is given to effects on working conditions, human health and violation of human, animal and/or nature rights.

Almost all valuation methods developed in connection to LCA involves a weighting procedure. It may be argued that already by taking the decision to perform a full LCA, a weighting procedure has been accepted implicitly. This is because if some aspects were given absolute priorities, it would normally be enough to consider these aspects and needless to perform a full LCA (Heijungs et al, 1992). Single-indicator methods based on e.g. mass displacement (Schmidt-Bleek, 1993) or energy requirements can possibly be seen as valuation methods, without weighting, focusing on a specific parameter.

LCAs will however in most cases include several types of parameters. The rest of this report will therefore concentrate on valuation methods including weighting procedures.

Although utilitarianism is much debated and controversial (e.g. Smart and Williams, 1973), expected utility as a decision criteria is the major paradigm in decision theory both in descriptive and normative applications (Hansson, 1991). It is sometimes suggested to be the "objective" decision criteria which in relation to this discussion can be seen as a clearly normative standpoint.

2.3 The weighting method

The choice of a weighting method may also be influenced by ideological and ethical standpoints. Here, three aspects will be discussed: views on the society, ethical views and views on nature.

2.3.1 Views on the society

The preferred weighting method, and aspects of the weighting method, may depend on views of the society and ideological standpoints. Much of the political debate during the 20th century has centred around views of the societal economy and representative democracy. These issues may also reflect choices concerning weighting methods in LCA.

Views on market economy
For people holding a positive view of the market economy as a system for exchanging information, it may be reasonable to advocate weighting methods which are using market prices and other types of information derived from the market. Such methods will typically result in weighting factors expressed in monetary units. However as discussed below, all methods using monetary units will not be based on values derived from a market. For persons holding a negative view of the market economy, information from the market is more or less useless. Persons with such views would probably not advocate a method based on market information. A conditional view of the market economy, suggesting that some information can be derived from the market in some cases, is an often encountered standpoint in many West-European countries.

Views on representative democracy
If it is assumed that decisions taken by democratically elected governments will represent the views of a society, and that valuation weighting factors should reflect these views, the weighting methods may ideally use information derived from governmental decisions. These types of methods may also result in weighting factors expressed in monetary units, if the weighting factors are derived in such units. For persons holding a

less positive view of representative democracy, decisions taken by governments may be less representative of the views of the society.

Views on Platonic philosophers/experts

Plato did not believe in democracy. He thought that if societies were ruled by philosophers/experts, better decisions would be taken. People subscribing to a similar view may suggest that difficult decisions should be taken by experts and not by common people or their representatives. For people holding these views, it seems reasonable that weighting factors may be derived from the opinions of experts. For other people, this view is undemocratic. They would perhaps suggest that there is a role for experts in providing the basis for decisions but when ideological decisions are taken, there is no special role for experts. Weighting factors derived from a panel of "Platonic experts" only representing themselves, would then not be authoritative. (Such a panel should however not be mixed with a panel consisting of representatives from different stakeholder groups. Such a panel may also be called an "expert panel", especially if different stakeholders choose to be represented by "experts". However, from a democratic standpoint, such a panel may be representative and thus authoritative for some people).

2.3.2 Ethical views

Something that is valuable, may either have an instrumental value or an intrinsic value to somebody. If something has an instrumental value, it is valuable because it can be used in order to gain something that has an intrinsic value. Something that has an intrinsic value, is valuable in itself. Both instrumental and intrinsic values are valuable in relation to somebody or something performing the valuation. If this "somebody or something" performing the valuation has an ethical value we may call this "somebody or something" a moral object (Ariansen, 1993). If we are following a universal moral, we have at least to some extent, ethical obligations towards moral objects. It is generally accepted that living people are moral objects. One interesting question is however if the class of moral objects should be expanded beyond that. Another important question is what relative importance different moral objects should be given.

Are all living people equally important?

A positive answer is in line with e.g. United Nation's declaration of human rights and it is a principle which is generally accepted in most liberal democracies. However, people and governments are perhaps not always acting according to this principle. If weighting factors are derived from the behaviour of people or governments, weighting factors may thus be in conflict with the principle. Also if willingness-to-pay measures are used without income adjustments, the resulting weighting factors may violate the principle. This is because the possibility-to-pay varies with income.

Are future people moral objects and if so, how important are they?

This question has a direct consequence on the methodological question how to handle future impacts in an LCA. Should future impacts occurring after a certain time period be cut-off and neglected? Should some sort of discounting be used?

A cut-off is consistent with a view that future people are not moral objects. The purpose of discounting is to discriminate against the future (Turner et al, 1994). There may however be different reasons to do that. A discount rate is often described as consisting of two parts. 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). Often the growth 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 may thus be consistent with a view that future people are moral objects with the same relative importance as current people. In conclusion, the choice of discount rate is an ideological issue, both in relation to the ethical question on time preferences, and in relation to the expected growth in the future. Views on discount rates are also likely to be affected by views on the market economy since the rate may be seen as the market price for capital.

In most currently available LCA valuation methods, these questions are not explicitly addressed. This is for example the case if weighting factors are used in which an integration over time has already been done. For example, if a weighting factor is derived from a governmental decision on emission targets, already in the decision, an integration over time has been performed. The discount rate, or the cut-off, is then the same as was implicitly used when the decision was taken. Although the ideological choices in these cases are implicit, and not well described, they are never the less present.

A decision on cut-off or discount rates is however necessary if the ambition in the impact assessment is to describe the damages done by different interventions (i.e. emissions or resource use), followed by a valuation of the damage (see also chapter 3). In the EPS-system (Steen and Ryding, 1992), this is partly the case. No discount rates are used, instead the valuation is performed by integrating over a chosen time-frame. This time-frame varies. For most effects from global warming, an integration over 100 years is used (Steen, 1995). For heavy metals, effects on forest production is integrated over 100 years, whereas health effects are integrated over one year (Boström and Steen, 1994). In these cases, a cut-off is thus implicitly performed.

If future people are to be considered as moral objects, one problem comes from the fact that it is only current people that can be involved in the actual weighting. In the weighting method, there may thus be a need for some formal procedure to secure that the values of future generations are not neglected, to the extent that these are foreseeable.

Are also animals, plants, and/or ecosystems moral objects?
This is of course a fundamental ethical question in which different opinions may be encountered in the society. If the answer to any of the questions is yes, this may have some consequences for the weighting method. There may also in this case be a need for some formal procedure to secure that the values of all moral objects are considered, to the extent that we can understand these values.

Are equality and justice of importance?
Environmental risks are not distributed evenly across populations (Harding and Holdren, 1993). Often low-income groups receive disproportionate shares of environmental hazards. The question of justice is related to what is considered "proportionate" shares, (proportionate in relation to what?). Important ideological questions are then, what is the just distribution and how important is it that it is fulfilled? When discussing life cycle valuation methodology, these questions may be summarised as: Does it matter who is affected by the environmental impacts? In currently available weighting methods, no consideration is given to these questions. Thus in practise, environmental justice is not given any consideration.

It may be argued that equality and justice are aspects that are difficult to handle in a Life Cycle Impact Assessment. This is because it in general is not possible to handle site-specific aspects (Udo de Haes, 1996). However, in a generic site-dependent approach, questions related to equality and justice could possibly be included. To be able to handle these questions, a further development of LCA in general may thus be needed.

2.3.3 Views on nature

To what extent are we able to predict environmental impacts?
The answer to this question is dependent on our views on nature. Nature may for example be seen as benign or surprising (Wiman, 1990). A benign nature will respond when exposed to stress, then if the stress is lessened or removed, nature will adjust it self to the former state of behaviour. A surprising nature may hide the response when exposed to stress, then at some time flip to another state in a more or less irreversible way. Our possibilities to predict environmental impacts will be larger if nature is benign rather than surprising. These questions are also linked to the precautionary principle and its application. Assuming a benign nature, the precautionary principle may be seen as unscientific and unnecessary. Assuming a surprising nature, the precautionary principle

is probably necessary. The answers to the question in the heading will also influence the choice of methodology for deriving weighting factors in valuation methods. If we to a large extent are able to predict environmental impacts, it may be of interest to look for valuation methods in which the environmental damages are predicted and valued. If we to a large extent are unable to predict environmental impacts, it is reasonable to look for valuation methods in which it is not necessary to evaluate the environmental damages, since this to a large extent will be impossible.

2.4 The weighting factors

After a weighting method has been chosen, also the weights themselves will be influenced by ideologies. Below, some aspects will be briefly discussed. Some of these were already mentioned above as ideological views on the weighting methods.

Views on market economy

For persons holding a positive view on market economies, environmental assets which already have a place on the market are likely to be valued as being less important. An example could be non-renewable resources which some people argue is not a problem. This may be done from the view point that the market will see that new resources are developed when needed (e.g. Dasgupta, 1989).

Also, a positive view on market economy may suggest that market derived discount rates are used, resulting in less weight to impacts occurring in the future. For persons with these views, a concentration on impacts occurring in the near future, with external effects which the market will not handle is probably of more relevance.

Are future people moral objects and if so, how important are they?

Views on the importance of future people will influence the weighting of impacts in the distant future compared to impacts in the near future.

Are also animals, plants, and/or ecosystems moral objects?

Animals, plants and ecosystems have an instrumental value for humans. If they also have an intrinsic or inherent value, it is likely that impacts on them are weighted more heavily than if only the instrumental value is considered.

To what extent are we able to predict environmental impacts?

Persons holding a more negative view on our possibilities to predict environmental impacts are likely to stress the importance on the precautionary principle. They will probably give a greater weight to impacts where larger uncertainties prevail compared to more well studied impacts. Examples of impacts which larger uncertainties are impacts occurring in the distant future, impacts from less studied chemicals, and impacts on biodiversity.

<u>What is the importance of the natural systems in relation to the economic systems?</u>
The overall economic system is an open subsystem of the overall ecosystem (Folke, 1990). Economic systems use ecosystems as sources for energy and natural resources and as sinks for waste. The economic systems are also dependent on a number of environmental services provided by the ecosystems. Different persons may have opposing views on how important the overall ecosystem is for the overall economic system. Persons emphasising the importance of the ecosystem are likely to weigh impacts on ecosystems and their functioning more heavily.

<u>What is the long-term development of natural systems?</u>
Two extreme position may be taken in relation to this question (Wandén, 1993). According to the first position, nature will develop into a climax situation. According to the second position, nature will always change and develop and it is not possible to compare different situations. The first position is often accompanied by a valuation of the climax situation as something intrinsically valuable. Man should try to avoid disturbing the nature in order to let the climax situation be developed. The second position is often accompanied by a standpoint that since different situations can not be compared, there is no intrinsic difference between systems influenced by man and not. Persons taking the first position are likely to value undisturbed ecosystems and biological diversity more than persons taking the second position.

2.5 Conclusions

As discussed above there are several ethical and ideological standpoints that will influence not only the valuation weighting factors, but also the choice of weighting method and the choice whether a weighting method should be used at all. These ethical and ideological standpoints are in most cases not explicitly discussed in the development of valuation methods and weighting factors. It may therefore be easy to forget them and believe that they are not there. It is also often difficult to understand which positions that have been taken in different methods. However, in the choices made, these ideological standpoints are taken implicitly. Since different persons do have different ideologies, we can expect differences, not only in weighting factors, but also in preferred weighting methods, giving different results. The differences in the results will remain difficult to understand and explain as long as the ideological standpoints are taken implicitly.

3 The framework - What is valued?

3.1 Introduction

The framework described in chapter 1 has been developed during the 1990s and will here be called the SETAC framework. It is largely the same as in SETAC's 'Code of Practise' (Consoli et al, 1993). Although it is widely accepted in general, there are still ongoing discussions and there may be some changes for example in the final ISO standards.

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 does 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 view-point 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 (sub)components were relevant. Examples of projects which followed this path was the development of the Dutch guidelines (Heijungs et al, 1992) and the 'LCA-Nordic'-project (Anonymous, 1992, Lindfors et al, 1995b). 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 (e.g. Kortman, et al, 1994, Kalisvaart and Remmerswaal, 1994). It can therefore be argued that it is still an open question to what extent the SETAC framework is useful when the aim is to develop valuation methods.

3.2 The cause-effect chain

The cause-effect chain of environmental problems can in simplified terms be schematically described as in Fig. 3.1 (Finnveden et al, 1992). Different activities causes emissions of various substances. The emissions lead to increased concentrations of these substances in the environment. The substances cause changes in the environment, which can be called the primary effects. Different substances can contribute to the same kind of change and one substance can contribute to several kinds of primary effects. The primary effect can then be the cause of a new kind of change, which can be called the secondary effect. One primary effect can be the cause of several secondary effects and vice versa, several primary effects can be the cause of one secondary effect. This cause-effect chain is then continued and is in principle never-ending. There are also possibilities for feedback mechanisms. Higher order effects can be the cause of lower order effects and emission of substances.

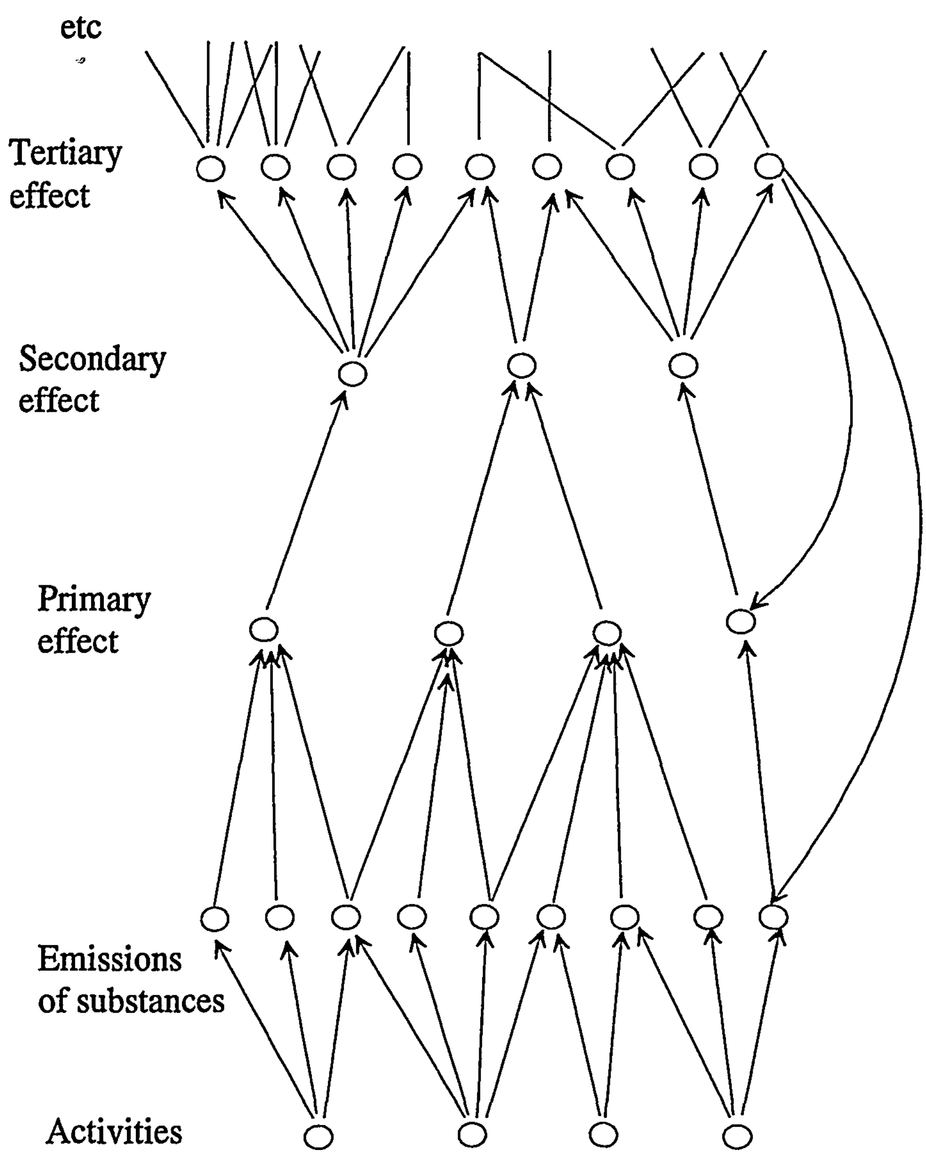

Fig. 3.1 A schematic description of the cause-effect chain (Finnveden et al, 1992).

As an example consider "Global warming". This impact is directly caused by several compounds. They all absorb infra-red radiation and this leads to a disturbed balance between the energy absorbed by the earth and the energy emitted by it. This change of "the radiative forcing" can be called the primary effect. The change in radiative forcing is expected to change the global temperature and this could be called the secondary effect. The temperature change is expected to lead to tertiary effects such as ice-melting, raising of the sea-level and other changes of the whole climatic system. These changes will in turn cause higher-order effects such as changes in different ecosystems, impacts on human health etc. Besides compounds contributing directly to global warming, there are also a number of compounds that can contribute indirectly. There are also a number of feed-back loops complicating the picture.

In a characterisation, the contribution to an impact from an environmental intervention is to be quantified. This means that the impact has to be defined somewhere in the cause-effect chain. In general terms a distinction can be made between defining the impact early or late in the cause-effect chain. Earlier in the chain, the effects will often be chemical or physical. Later in the cause-effect chain, the effects are more often biological changes. In general, the uncertainty will be larger later in the cause-effect chain. This is perhaps the reason why most characterisation methods have defined the effects early in the cause-effect chain rather than late.

As briefly discussed in chapter 2, different people will have different opinions on what has intrinsic value. However, for most people things that have intrinsic value will in general be found later in the cause-effect chain. Environmental interventions and effects earlier in the cause-effect chain will then have an instrumental value (usually negative) with respect to the things that have intrinsic value.

3.3 Environmental threats and Areas for protection

In the classification, the inputs and outputs of the system are assigned to different impact categories. An important step is then to decide which environmental impacts to consider in the impact assessment. There has been a considerable amount of discussions concerning this and different lists has been suggested (e.g. Consoli et al, 1993, Heijungs et al, 1992, Lindfors et al, 1995, Udo de Haes, 1996). Despite the discussions, the different lists has many aspects in common. In Table 3.1 is the checking list suggested in the Nordic Guidelines presented as one example.

Table 3.1. List of impact categories, the categories can be further divided into subcategories (Lindfors et al, 1995).

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.

It can be noted that most impact categories in Table 3.1 relate to effects early in the cause-effect chain. Thus many of the impact categories are more related to negative instrumental values than intrinsic values. These impacts may be called "Environmental threats", they are threatening intrinsic values.

In SETAC's 'Code of Practise' three "Areas for protection" are defined: Resources, Human health and Ecological health (Consoli et al, 1993). The Areas for protection can be interpreted either as intrinsic values or as instrumental values closely connected to the intrinsic values. Impacts on the Areas for protection is thus usually found later in the case-effect chain. Although the classification and characterisation are most often performed on impacts early in the cause-effect chain, it can in principle also be performed later in the cause-effect chain and relate to "Areas for protection" rather than "Environmental threats".

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 an "Area for protection-based classification" the emissions are thus followed much longer in the cause-effect chain compared to an "Environmental threat-based classification". As an example, substances contributing to global warming are in the latter case normally compared on the basis of their relative impact on the radiative forcing, a physical effect early in the cause-effect chain. In the EPS-system, the effects on biodiversity, production and human health from, for example, the emission of CO_2, should in principle be calculated. Thus a classification/characterisation on the levels of "areas for protection" require an estimation of the damages.

3.4 Discussion

Consider the cause-effect chain. In the beginning of it, are the data from the inventory analysis. In the later parts of it are the intrinsic values. In between are impacts on the level of Environmental threats. The Areas for protection may be the intrinsic values or instrumental values close to the intrinsic values.

The choice of the intrinsic values, and the comparison between them, is as discussed in chapter 2, an ethical and ideological valuation. The relationships between an emission

and the potential impacts, may it be on the level of "environmental threats" or on the level of "areas for protection", is in principle a question to be answered by natural sciences. Thus in going from inventory data to a valuation, there is both natural science based information and valuations to be considered. From a methodological standpoint, the interesting questions are now:
* What type of information should be valued?
* Where in the cause-effect chain should the classification/characterisation be placed?
* Should the valuation be based on data from the inventory analysis, on an "environmental threat"-based classification/characterisation or on an "area for protection"-based classification/characterisation?

In general the uncertainty in the natural science based information will increase along the cause-effect chain. Thus, if the characterisation is placed close to the inventory data, the uncertainty in the characterisation results will generally be lower than if the characteisation is placed later in the cause-effect chain. On the other hand, if the characterisation is placed close to the inventory data, the uncertainty in the natural science based data may instead have been moved to the valuation.

The answers to the questions above will depend on the answer to the question: To what extent are we able to predict environmental impacts? This question was identified in chapter 2 as one of the ideological questions influencing the choice of valuation methodology. Persons with a positive view on our abilities to predict environmental impacts will perhaps suggest that the classification/characterisation is put close to the intrinsic values, persons with a less positive view will suggest that the classification/characterisation is put closer to the inventory data.

Our abilities to predict environmental impacts will also vary with different impacts. For impacts like acidification and eutrophication which has been known environmental problems for decades, and for which there is a lot of information available (although there of course are data gaps also), predictions may be possible. If, on the other hand the focus is on new chemicals for which only a few physico-chemical data is available, it is difficult, if not impossible, to predict the environmental impacts.

The difficulties of predicting impacts from organic chemicals is illustrated in the EPS-system. It was concluded by 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 most well studied groups of organic chemicals, this means that it will be virtually impossible to include any hazardous organic chemicals in the EPS-system.

Where to place the classification/characterisation in the cause-effect chain will also depend on the valuation method since different types of methods may require different types of data.

3.5 Normalisation

As noted in chapter 1, an additional subcomponent "normalisation" is sometimes suggested. Although there may be several uses for a normalisation step, the main purpose is usually to facilitate the valuation.

"Normalisation" is often defined as a step in which the data from the characterisation are related to (i.e. divided by) the total magnitude of the given impact category in some given area and time (Consoli et al, 1993). The "given area" may for example be the globe, a region or a country. The time aspect refers to the actual present or past contribution, or a predicted future contribution. Possible extensions of the definition may include a normalisation on inventory data and also other types of reference situations. The latter may for example be a desired situation in contrast to an actual situation.

The possibilities for normalisation depends on the possibilities of calculating the total magnitude of the given impact category. This in turn depends on the possibilities of finding the necessary data for the given area and time. These possibilities will be different for different impact categories. For some, e.g. related to toxicological impacts and biological diversity where a large amount of different intervention parameters will contribute to the impact, it will be difficult, if possible at all, to calculate the total contribution (Finnveden, 1994, 1996).

Normalisation means a further manipulation of data, and an introduction of new uncertainties. From this perspective, it will be an advantage if it can be avoided. If it on the other hand facilitates the valuation, this is an argument for doing it. It seems like for some types of valuation methods, it is a necessary prelude to valuation. This will be further discussed in the next chapter.

4. Valuation methods - How is the valuation done?

4.1 Introduction

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). From a methodological standpoint, this distinction is however of limited importance since essentially the same methods can be used in both cases. The disctinction is however relevant from another standpoint. In chapter 1 it was noted that a distinction can be made between the use of valuation methods and drawing conclusions. If a case-specific weighting set is used, derived for the specific case, the difference between using valuation methods and drawing conclusions 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.

The quantitative valuation methods may be divided into three main groups (based on Lindeijer, 1996):
1. Panel methods
2. Monetarisation methods
3. Distance-to-target methods
Combinations of these methods are also possible.

4.2 Qualitative methods

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 4.1.

Fig. 4.1 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, the severeness of an impact is indicated by a sign or a number. 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 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), 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 4.2.

Fig 4.2. 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.

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.

4.3 Quantitative methods

All quantitative methods discussed in this chapter 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

$$EI = \Sigma V_i I_i$$

where I_i is 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.

4.3.1 Panel methods

There are many similarities between qualitative methods and quantitative panel methods. In both cases a group of people are asked about their opinions. The major difference is that in the latter case, people are asked to express their opinions in a quantitative way. Major methodological questions include how the questions are being formulated and asked.

Examples of studies in which quantitative panel methods have been used are two different Dutch studies (Anonymous, 1991 and Kortman et al, 1994) and a British study (Wilson and Jones, 1994).

In the first Dutch study (Anonymous, 1991, Annema, 1992) the starting point was normalised characterisation results. The application of weighting factors were 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. This step thus resemble the determination of the "ecological importance" in the German qualitative system described above. The second step of the process was a first assessment of the weighting factors. This step was confidentially done by each member. In the third step, the results was 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 the 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. 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 (see below) were presented to the experts which 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 were 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.

These three examples illustrate different approaches for panel methods. Other techniques used in decision theory which has been discussed in connection with LCA are the multi-attribute utility theory (MAUT) and analytical hierarchy process (AHP) (e.g. Fava et al, 1993). These techniques may be useful to help a person structure and formulate his or her opinions.

4.3.2 Monetarisation methods

At first glance monetarisation methods may seem very different from panel methods. This may also be the case but not necessarily. In the methods discussed above, people are asked to distribute points on different items. Very similar methods may also be used with the difference that people are asked to put monetary values on the same items. Such methods would be called monetarisation methods, but could equally well be called panel methods.

There is a large number of different approaches for monetarising 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 (no distinction is made here between "willingness-to-pay" and "willingness-to-accept").

<u>1. Methods based on willingness-to-pay.</u>
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. With reference to the discussion on cause-effect chains in chapter 3, this something may be early or late in the cause-effect chain. 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 <u>user values</u> and <u>non-user values</u>. The user values include both <u>direct and indirect user values</u>. An example of a direct user value is the timber value of a forest. The indirect user value include the recreation value of the forest, the value of carbon fixation etc. (Turner et al, 1994). The non-user value include option use value, bequest value and existence value. The total economic value is the sum of the user and non-user values.

A number of different methods may be used to derive a willingness-to-pay measure. Here will a distinction 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.

1.1 Individual's revealed preferences

Methods based on individual's 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 user values, and sometimes only the direct user value. Direct user values can often be derived from actual market prices, e.g. the market price of timber. Also the indirect user values may be derived from market values, though often indirectly. Examples of methods for evaluating total user values are the travel cost method and hedonic pricing methods. The travel cost method is a revealed preference method which 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 which are determined by a number of factors, including environmental aspects. Another example is wages which may vary depending on the risks associated with different types of jobs.

1.2 Individual's expressed preferences

Non-user 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 above. In both types of methods, 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).

1.3 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 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.

2. Methods not based on willingness-to-pay

There are also a number of monetarisation methods which 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 the approach mentioned above, in which 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.

In the LCA-world, there is currently a couple of valuation methods suggested which are based on monetarisation. In the EPS-system (Steen and Ryding, 1992), the valuations are based on different types of measures. 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 (heading 1.1 above) and contingent valuation methods (heading 1.2 above). Biodiversity is valued per capita as 10 times the amount the Swedish government is spending on conservation of biological diversity. Production losses is valued from market prices. Resources is valued based on assumed future costs (Steen, 1995). Different types of monetarised measures are thus used for different problems.

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 and marginal costs for reducing emissions down to decided emission limits.

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.

4.3.3 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
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

$$V_i = \frac{1}{T_i} \tag{4.1}$$

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 4.1, 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 procedure as discussed in section 3.5. (It will in fact be a normalisation procedure if a wide definition of "normalisation" is used to include also "target levels" as "reference levels"). This equation was used by Schaltegger and Sturm (1991). The target levels was different types of quality standards, expressed as amount per mole of media (air and water). The valuation is performed on data from the inventory analysis.

Equation 4.1 may also be written as

$$V = \frac{1}{A}\frac{A}{T} \tag{4.1'}$$

where A is the actual flow within a specified area and time. The factor 1/A is thus the normalisation factor as discussed in chapter 3.5. Equation 4.1' 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 an "environmental threat"-based classification/characterisation. The targets were either Swedish short-term political targets or long-term "critical levels".

Essentially the same equation (4.1') is used in the MET-points method (Kalisvaart and Remmerswaal, 1994). The targets are Dutch environmental policy goals for year 2010. The valuations is performed on results from an "environmental threat"-based classification/characterisation.

A development of equation 4.1 is to include a subjective weighting factor, W:

$$V = W\frac{1}{T} \qquad (4.2)$$

Equation 4.2 is identical to 4.1 if the subjective weighting factor is put to 1 for all interventions. Equation 4.2 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 an "Environmental threat"-based classification/characterisation.

Equation 4.2 was also used by Goedkoop (1995) in the Eco-indicator 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 an "environmental threat"-based 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 here is 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.
The valuation in this case is thus performed on results from an "Area for protection"-based classification/characterisation.

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)

$$V = \frac{1}{T}\frac{A}{T} \qquad (4.3)$$

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.

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

$$V = W\frac{1}{A}\frac{A\text{-}T}{T} \qquad (4.3)$$

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).

4.3.4 Combinations of methods and a brief discussion

In the distance to target methods, the subjective weighting factor is normally set to one which means that all targets are arbitrarily assumed to be equally important. In a sense it can therefore be questioned whether distance-to-target methods in general can be called valuation methods (see also next chapter). A second method, which can be either a panel or a monetarisation method, is required in order to calculate the subjective weighting factor. Another approach could be to use a panel to set targets in such a way that they can be assumed to be equally important.

The distinction between panel, monetarisation and distance-to-target methods is useful for structuring the discussion. In some ways it may however also be misleading. There are for example similarities between some monetarisation methods, notably the contingent valuation method, and panel methods, since in both approaches individuals are asked about there preferences. There are also similarities between some distance-to-target methods (those that are based on political/administrative targets) and those monetarisation methods which are based on society's willingness-to-pay since these are also based on political/administrative decisions.

A possible other way of classifying different valuation methods, in a second dimension, could be based on who is doing the valuation. Two broad classes can then be defined:

1. Valuation methods based on individual's preferences
2. Valuation methods based on society's preferences as derived from political and administrative decisions.

The first class can the be divided into several subclasses

a. Individuals representing an entire population
b. Individuals representing different subgroups of the entire population
c. Individuals representing stakeholders
d. Individuals representing themselves as experts

In all these cases can individual's preferences be evaluated either through panel methods, contingent valuation methods or from revealed preferences in a market situation.

5 Presently available valuation methods

5.1 Results from some comparative studies

One starting point for a discussion on valuation methods is results from case studies in which several methods have been used and the results compared. In one of the case studies used in the development of the Nordic Guidelines (Lindfors et al, 1995), several valuation weighting sets were used:
* The EPS-system, data mainly from Steen and Ryding (1992).
* The Tellus-system, data mainly from Tellus Institute (1992).
* Ecoscarcity, Swiss data from Ahbe et al (1990).
* Ecoscarcity, Swedish data from Baumann et al (1993).
* Effect-category, short time, data from Baumann et al (1993)
* Effect-category, long time, data from Baumann et al (1993)
* The "molar method", data from Schaltegger and Sturm (1991).

In Table 5.1, the results for those parameters that contributes to more than 10 % to the overall result according to at least one method are shown as contribution in % to overall result.

Table 5.1. Valuation results from the energy base case study as contribution in % to total result (Lindfors et al, 1995).

	EPS	Tellus	Ecoscarc, Swiss	Ecoscarc, Swedish	Effect.cat short	Effect.cat long	Molar method
Raw materials							
Coal	32	-	5	0,3	5	5	-
Air emissions							
CO_2	44	8	10	0,3	19	33	2
NO_x	0,4	26	44	0,4	26	13	49
SO_2	0,1	30	17	0,3	11	8	34
TSP*	0,005	16	-	-	0,09	0,05	8
VOC	-	2	2	88	2	3	0,01
Water emissions							
Oils and greases	-	-	-	-	1	17	0,005
Solid wastes							
Other industrial	0,0003	-	14	1	14	8	-
Remaining	24	18	8	10	22	13	7

It can be noted that none of the parameters are of major importance according to all methods. Emissions of CO_2, NO_x and SO_2 are however of importance according to most methods. It is also the only three parameters for which there are weighting factors available for all methods.

In Braunschweig et al (1994) a comparative study is performed on a "Global Annual LCA", i.e. data corresponding to environmental interventions for the whole earth was used. The authors however stress that the data is uncertain to an unknown extent and that the results should be interpreted carefully. Nevertheless it is an interesting idea to compare different valuation methods on "global data". In Table 5.2 are some of the results summarised, again only showing those parameters that contribute to at least 10 % according to any method. The following methods are compared in Table 5.2:
* NSAEL-method (No Significant Adverse Effect Level), the distance to target method developed by Kortman et al (1994).
* PANEL-method, the panel-method developed by Kortman et al (1994)
* MET-method, the distance to-target method, developed by Kalisvaart and Remmerswaal (1994).
* Ecoscarcity, Swiss data as described in Ahbe et al (1990) and Braunschweig and Müller-Wenk (1993).
* EPS-method as described in Steen and Ryding (1992)
* Landbank, the panel-method developed by Wilson and Jones (1994)

Table 5.2 Valuation results from a "Global LCA" as contribution in % to total result. (Braunschweig et al, 1994).

	NSAEL	PANEL	MET	Ecoscarcity (Swiss data)	EPS	Landbank
Resources						
Oil	-	-	-	-	11	-
Energy	-	-	-	2	>21 (sum of oil, coal and fossil gas)	-
Silver	-	-	-	-	26	-
Platinum	-	-	-	-	10	-
Air emissions						
CO_2	5	17	4	6	19	87
NH_3 (both air and water)	12	6	4	not shown	-	-
NO_x	17	11	8	28	0,2	1
SO_2	27	15	11	20	0,1	2
CFC-11	6	7	11	8	1	-
CFC-12	8	10	14	not shown	not shown	-
CFC-11-eq. (including CFC-11)	not shown	not shown	not shown	27	3	-
Nitrate	12	12	9	-	-	-
Remaining	13	22	39	9	21	10

In this case, four of the methods give rather similar results. NSAEL, PANEL, MET and Ecoscarcity all indicates that nitrogen-emissions to air is the most important single emission. Also SO_2 and CFCs are of importance as well as CO_2 to a somewhat lower and varying extent. This is in contrast to the EPS-system which finds that the most important aspect is depletion of non-renewable resources, both energy and metals, especially Ag and Pt. Also CO_2-emissions are of relevance. The Landbank panel gave highest priority to CO_2-emissions. It should be noted that water emissions are not given any priority according to any of these methods.

From these comparative studies it is clear that different methods will give different results. There is thus a possibility to choose methods according to different ideological standpoints. For example, those that believe that depletion of non-renewable resources is a very important problem together with global warming should choose the EPS-system. Those that do not consider depletion of non-renewables to be a problem, but consider Global Warming to be the largest problem should choose the dataset developed by Landbank. Those that consider the traditional air pollutants (NO_x, SO_2 etc.) to be of largest significance can choose essentially any of the usual distance-to-target methods. Those that are interested in specific potentially hazardous chemicals may find a problem to find a valuation method reflecting these values. The method from the Tellus Institute can in some cases (not shown here) give at least some priority to hazardous chemicals. Those that think water pollution is important will have troubles finding a method reflecting this standpoint.

5.2 Discussion

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 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.

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 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. 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. By assuming that all targets are equally important, the distance-to-target methods are in a sense avoiding the explicit weighting. It can therefore be questioned whether the distance-to-target methods are valuation weighting methods at all. They can also be seen as extended normalisation methods.

One argument that sometimes is raised in favour of a distance-to-target method based on political/administrative targets is that the valuation methods should be based on the societal goals as expressed by democratically elected bodies. This argument may also be used pro monetarisation methods in which the monetary values are derived from political/administrative decisions. In these monetarisation methods not only the targets may be considered, as in the distance-to-target methods, but also the importance of the different targets if it can be assumed that the monetary value is somehow related to the importance.

If it is assumed, that environmental policy goals at least implicitly are set from two perspectives: the environmental importance and the costs of attaining the goals, then the target reduction levels will be a function of the importance and the inverse costs. If so the environmental importance will be a function of the targets and the costs of attaining the targets. This approach may be seen as a further development of a distance-to-target method with a monetarisation part to evaluate the importance of different targets. This approach seems to be largely the same as the DESC-method (Krozer, 1992) however the publicly available description of the method is apparently very limited and prohibits a further discussion of it.

The discussion above raises the question: on what basis are environmental policy goals set? This is a question which can be studied in political sciences and it may be possible to find support for different assumptions in that area.

Another monetarisation method which is also based on political/administrative decisions is the Tellus method (Tellus Institute, 1992). It only includes some types of impacts, there is thus significant data-gaps. However, in contrast to many other valuation methods, it includes data for substances with toxicological relevance. These have been calculated based on a specific characterisation method for human toxicological impacts. This characterisation method, in parallel to others for human health impacts, are currently being discussed (see e.g. Lindfors et al, 1995). The valuation weighting factors are however in principle open for recalculation using other human health characterisation methods.

The EPS-system is unique among valuation methods in including valuation weighting factors for other resources than energy. As noted above, non-renewable resources will in most case studies turn out to be very important for the final result. The method by which different non-renewable resources are weighted against each other (this part may be seen as a characterisation method) is quite different to other published methods for characterising non-renewable resources, giving different results (Lindfors et al, 1995). In the EPS-system the valuation weighting factors are calculated from estimations on costs for production processes which are assumed to replace the current processes in the future. The results will of course depend on the assumptions made when designing the future processes.

The valuation of emissions are in the EPS-system based on estimations of damages. A necessary requisite is thus that estimations of damages can be made. If this is not possible, no index can be calculated, in apparent conflict with the precautionary principle. As noted in section 3.4, this will be the case for most organic chemicals.

In all valuation methods there are data gaps in the sense that weighting factors are lacking for a number of relevant parameters. In the EPS-system there is also another type of data gap in the sense that when a valuation weighting factor for an emission parameter is available, not all relevant types of environmental impacts which this pollutant may contribute to, may have been included. The calculation of the index for cadmium will here be used as an example (Boström and Steen, 1994). Cadmium may cause both human toxicological as well as ecotoxicological impacts. In relation to the latter, only production loss in forests due to decreased microbiological activity is considered by Boström and Steen. With relation to other types of ecotoxicological impacts, there are thus data gaps. In relation to human toxicological impacts, only exposure via inhalation is considered. Other major exposure routes, via e.g. food (OECD, 1995) is not considered in the EPS-system, resulting in data gaps. These data gaps are however not apparent. In order to identify them, a thorough review of the background material, in combination with a 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 has apparently never been performed.

A number of reviews of valuation methods has been published by LCA experts (this is yet another one). One of the more comprehensive (Braunschweig et al, 1994) points to the possible logical contradictions in the EPS-system in which parameters which may have different units are added.

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 recent 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. This suggests that there is a systematic difference in the calculations and that either may be subject to large errors. This is however not further investigated here.

One of the lessons from recent 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 is 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.*).

One of the few studies in which panels have been used and the weighting factors have been published is described by Wilson and Jones (1994). The weighting factors derived by the panel is 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.

6 Valuation weighting methods in relation to Applications

6.1 Introduction

When discussing different uses of LCA, this can be done in several dimensions (see also Udo de Haes, 1993).

A first dimension is the user of an LCA. Here can a distinction be made between governments, companies or NGOs. These main actors can be further divided into subgroups. For example, it seems likely that different types of companies will use LCAs differently. Producing companies may for example be divided into five groups related to their driving forces (Omrcen, 1995):
1) Product-driven companies have their main driving force in current products. The key competences are product development and marketing/service.
2) Market driven companies have their key competencies directed towards identifying customer needs and strengthening these ties by means of market investigations or needs analysis, as well as the ability to build up customer loyalty.
3) Technology-driven companies have their key competencies in research and development as well as marketing of applications.
4) Production-driven companies have production resources as driving force. Key competences are production efficiency and marketing of substitutes.
5) Natural-resource driven companies have their choice of product controlled by the raw materials or other types of natural resources that the company has at its disposal.
Within companies, there are also different actors; e.g. designers, environmental experts etc.

A second dimension relates to if the study is used externally or internally with respect to the funding organisation.

A third dimension concerns if the study is
* directly intended for decision-making
* intended for other purposes
Examples of the latter are educational purposes, as a tool for structuring information etc.
If the study is intended for decision-making, distinctions can be made between different types of decisions, again in different dimensions: decision can be (e.g. Baumann, 1994):
* strategic
* operational
Another dimension concerns the time-frame of the decision (Wenzel, 1994), e.g.
* short time-frame, (years)
* long time-frame, (decades)

* very long time-frames, (longer)
Yet another dimension in relation to decisions can be
* important decisions
* less important decisions

The last dimension to be discussed here concerns the application. Several surveys have been performed during the last years asking companies, governments and NGOs about applications of LCA. Some of these results will be reviewed here.

In a Nordic study by Finnveden and Lindfors (1992), the most frequent application mentioned was "product and process development". However, industry representatives often mentioned "strategic decision making" as the most important area for themselves. Also LCA as a support for "marketing and general information to the public" and for supporting "decisions on buying" was quite often mentioned although not as the most important ones.

In the international survey by Ryding (1994), two forms of applications from the manufacturer's point of view were given highest priority, namely to "identify processes, ingredients and systems that are major contributors to environmental impacts" and to "compare different options within a particular process with the objective of minimising environmental impacts". Other applications that were regarded as important were to "provide guidance in long-term strategic planning" and to "help to train product designers". Looking at forms of applications for public decision-makers the one option that was regarded as most important were to "help to develop long-term policy regarding overall material use, resource conservation and reduction of environmental impacts and risks posed by materials and processes throughout the product life-cycle".

In the survey by Baumann (1994), the most frequently mentioned applications or objectives for using LCA in Swedish industries were (in order of frequency):
1) Analysis of the company's own product
2) To learn about LCA
3) In product development
4) External use (marketing, ecolabelling etc.)
5) For process improvement and optimisation
6) To chose between suppliers or raw materials
7) For educational purposes

In a survey by Vigon and Jensen (1995), the following application categories were noted in order of frequency:
1) Internal product development/improvement
2) External product claims
3) Broad screening

4) Technical policy support
5) Other

Although the results from these surveys are somewhat different, there are also common features emerging. In the next section some application areas will be discussed in more detail in relation to the need for valuation weighting methods. The focus will be on applications in which documentation is available from which conclusions in relation to valuation methods can be drawn.

Before discussing the need for valuation methods, one may however start with discussing the need for impact assessment in general. Why do we need an impact assessment at all? Why is the information provided by the inventory analysis not enough?

When an LCA is to be used as a basis for a decision maker, the impact assessment is useful in two slightly different ways :
1. Translation. The information provided by the inventory analysis is expressed in terms of inflows and outflows. Thus in order to get information on environmental impacts, the information from the inventory analysis on inflows and outflows has to be translated and interpreted into a form which is relevant and useful for the decision maker. What is useful and relevant will then depend on the decision maker and on the situation in which the decision is to be taken.
2. Aggregation. The numbers of parameters generated by the inventory analysis can easily become quite large. There may therefore be a need to reduce the number of parameters to be able to make an assessment. Again, the extent to which an aggregation is needed will depend on the decision maker and on the decision to be taken.

With respect to valuation weighting methods, important questions are then:
* Is there a need for a complete aggregation? Does the decision maker need a complete aggregation in order to be able to take the decision?
* Is there a want for a complete aggregation? Does the decision maker want to know about the trade-offs before a decision is taken?
* Is a complete aggregation possible? Is it possible to weigh different aspects against each other or must the search for alternatives be continued until an alternative which fulfils all criteria are found?

Below, some described applications of LCA as a basis for decision making will be discussed in relation to the need for valuation weighting methods.

6.2 Examples of LCA applications

6.2.1 Product development and improvement

Product development and improvement is an LCA application which has received a large interest. There are a number of projects in different countries developing handbooks, guidelines and methods to be used in connection with "life-cycle design", "environmental-sound product development", etc. It is however interesting to note that the role of LCA (and LCA valuation methods) is quite different in different projects. One reason for this may be that different types of producing companies have different wants and needs. Another reason may be that these concepts are still relatively new and untested. Different approaches are thus being developed by different groups and it is still too early to evaluate them.

The EPS-system has been developed as a part of the Swedish Product Ecology Project (Ryding et al, 1995). The starting point of the EPS-system has been the designer. It has been an expressed wish that the designer should perform LCAs. It should be possible to base choices concerning e.g. materials and design on LCAs. This of course puts high demands on software and databases. It was also assumed that a designer is incapable of handling multi-dimensional environmental information. Therefore a valuation weighting method performing a complete aggregation giving a clear recommendation was required. In this approach, the LCA is expected to be used as an operational tool by the designers themselves when e.g. choosing between different materials. In the Dutch Ecoindicator-project, the starting point has been very similar (Goedkoop, 1995).

A different perspective is used in another recently started Swedish project for Green Concurrent Development (Karlsson, 1995). The starting point here is existing practises and methods of development and design such as Concurrent Engineering. In this project an Environmentally sound product is a product which fulfils expressed and unexpressed environmentally related needs and wishes that a company's stakeholders puts on the product during its material life-cycle. Here it is thus explicitly stated that when valuations are to be performed, it is the values of stakeholders that are of interest. Different stakeholders may have different values. Thus, it may be necessary to include a number of different sets of valuations. In the project, LCA seems to have its major role on a screening level for identifying major problem areas. Based on this information, and other sources, indicators are defined. These are used for communication of policy, goals and criteria and for feed-back. In this approach, the need for valuation weighting methods with a complete aggregation is more limited since the end result of the LCA is information on important aspects of the LCA, to be used when designing indicators.

Other projects has taken an explicit starting-point in models for design and product development recognising that LCA may be used differently in different phases of the development. In a guideline for Life-Cycle Design, Keoleian and Menerey (1993) (see also Keoleian and Menerey, 1994, and Keoleian (1993)) describe a typical design project as beginning with a needs analysis, then proceeding through formulating requirements, conceptual design, preliminary design, detailed design and implementation. During the needs analysis, the purpose and the scope of the project are defined. Needs are then expanded into a full set of design criteria that includes environmental requirements. Design alternatives are proposed to meet these requirements. The development team then continuously evaluates alternatives throughout design. If studies show that requirements can not be met or reasonably modified, the project should end. Successful designs balance environmental, performance, cost, legal and cultural requirements. Critical decisions must be made when developing requirements and evaluating decisions. Finally designs are implemented after final approval and closure by the development team. A similar model for describing the design process is used in the Danish EDIP-project (Environmental Design of Industrial Products) (Wenzel et al, 1994).

In this model, LCAs may be used in essentially two different phases: either as a part of the needs analysis and during the formulation of requirements, or during the design phases (Keoleian, 1993, Wenzel et al, 1994). In the first phase, an LCA is performed on one or several reference products. The aim is to find parts of the life-cycle responsible for major impacts, areas where improvements are possible. Based on the outcome of the LCA, design requirements can be formulated. In this type of LCA, the need for valuation weighting methods is limited, since the result of the LCA is a list of important aspects of the LCA which can be used when formulating design criteria.

During the actual design, LCAs may be used to evaluate different design concepts, choose between different materials etc. The evaluation of the LCA results can in this case be done in relation to the formulated design requirements. In such cases, no specific valuation method is required. There may however also be a need for a highly aggregated valuation weighting method in performing daily choices if the design requirements are not sufficiently specific or to solve less important trade off situations.

Other projects developing methods for product or material development have concentrated on qualitative or semi-quantitative methods using matrixes, check-lists and flagging criteria to identify parts which need further consideration (e.g. Graedel and Allenby, 1995, Schmidt et al, 1994). In these methods, the assessment is often done in close co-operation between designers and environmental experts. Reasons for concentrating on qualitative assessments include costs and difficulties in doing a full, quantitative LCA. Another important reason may be the impossibility of doing a full quantitative LCA early in a project where many details are still open or completely unknown, but where there still is a wish to consider the environmental aspects of the basic ideas so far generated.

As noted above, a distinction can be made between strategic and operational decisions. Strategic decision-making procedures are closely linked to the goals of the organisation, while operational decision-making procedures assumes that the goal is already decided (Baumann, 1994). An operational decision-making procedure deals with the control of given activities, while a strategic decision-making procedures deals with what activities are to be undertaken (*ibid.*). Recurrent situations with similar decision-making procedures are usually of an operational kind (*ibid.*). In recurrent decision situations, i.e. an operational use, it may be both more possible and more desired to formalise the decision-making process. It may be possible since it is recurrent and thus possible to learn from earlier situations. It may also be desired, because reproducibility may be an important requirement. A more formalised decision-making process may require a formalised valuation weighting method.

The question whether LCA is and/or should be an operational or a strategic tool is one of the issues being discussed (Omrcen, 1995). So far, LCA has only rarely, if at all, been implemented as an operational decision-making tool. Thus, LCA as an operational tool is still largely a future vision.

Decisions may have different time-frames. Decisions on for example investments or design of long-lived products, may have a long time-perspective, e.g. decades, whereas other decisions have shorter time-perspectives, e.g. years. Does this influence the need for valuation methods?

When trying to answer the above question, it may be useful to look back in time. How would a valuation method developed 15 or 20 years ago have prioritised different environmental problems? We do not know the answer. It does however seem reasonable to assume that it would have been quite different from current methods with regard to for example depletion of stratospheric ozone and global warming. However, it is interesting to note that all major environmental problems that are discussed today, were known, at least as environmental threats, 10 or 20 years ago, although not all of them were prioritised.

When looking into the future, a possible conclusion may be that we do not know today exactly how environmental problems will be prioritised 15 or 20 years ahead of us. From that it seems reasonable that all environmental threats known today, should be taken seriously. This is because we do not know which of the environmental threats will develop into major environmental problems. If all environmental threats are to be taken seriously, the need for valuation weighting methods is limited when long-term decisions are to be taken. Instead, decision-makers should look for solutions in which one or several threats are reduced while at the same time not significantly increasing other problems (this criteria for an optimal solution is sometimes called a Pareto criteria, e.g. Turner et al, 1994)

It is suggested here that the tentative conclusion that decision-makers often should look for a Pareto-optimal solution is valid both in public and private decision-making. In the case of public decision-making, it is often noted that long-term solutions of one environmental problem can not be at the expense of increasing other problems (e.g. Naturvårdsverket, 1993).

To support the conclusion concerning private decision making, consider as an example a refrigerator producing company. Due to international agreements, CFCs has to be replaced by something else. The questions are: With what?, What are the consequences over the life-cycle? and Are these consequences acceptable? It is suggested here, that for an environmentally responsible company, it would in the long run not be possible to solve the CFC-problem, while significantly increasing another problem. It would for example, not be possible to replace the CFCs, with something that significantly increased the risks of explosions of the refrigerator. Neither would it be possible to replace the CFCs with something with other types of significant health hazards. A significant increase in electricity demands during the use phase would probably also be impossible. Thus, it seems like the search for replacements would have to be continued until an acceptable alternative has been found. The acceptable alternative would solve the CFC problem, while at the same time not significantly increasing another problem, i.e. a Pareto-optimal solution.

Summing up so far, in several applications areas, the LCA is expected to result in an identification of critical areas. Only in a limited number of applications is the LCA expected to result in a one-dimensional score. The need for a complete aggregation is therefore limited. It is expected that the decision-maker in many situations wants to know which the trade-offs are, before taking the decision. It is furthermore expected that in many situations trade-offs are not possible. There will often be a number of criteria that must be fulfilled. If an acceptable solution is not found, the search for other alternatives must be continued.

Although the need for valuation methods, may seem limited there are however several important applications for valuation methods. As noted above, there are LCA application areas in which there is a need for a one-dimensional score as a result. Although decision-makers, in many situation wants a disaggregated result from an LCA, the results from valuation methods may be useful in addition to the other types of results, Fig. 6.1 (Lindfors et al, 1995b). Yet another important application of valuation methods is as a part of the LCA procedure discussed in e.g. Lindfors et al (1995b), Fig 6.2.

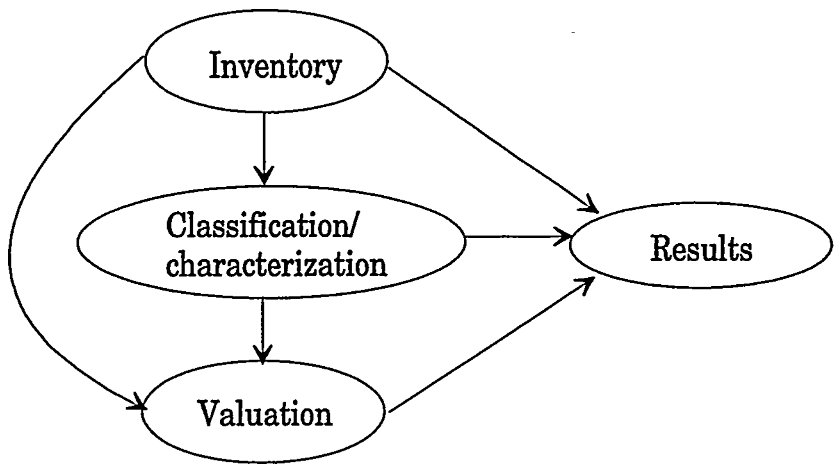

Fig. 6.1. Different types of results from a LCA (Lindfors et al, 1995b).

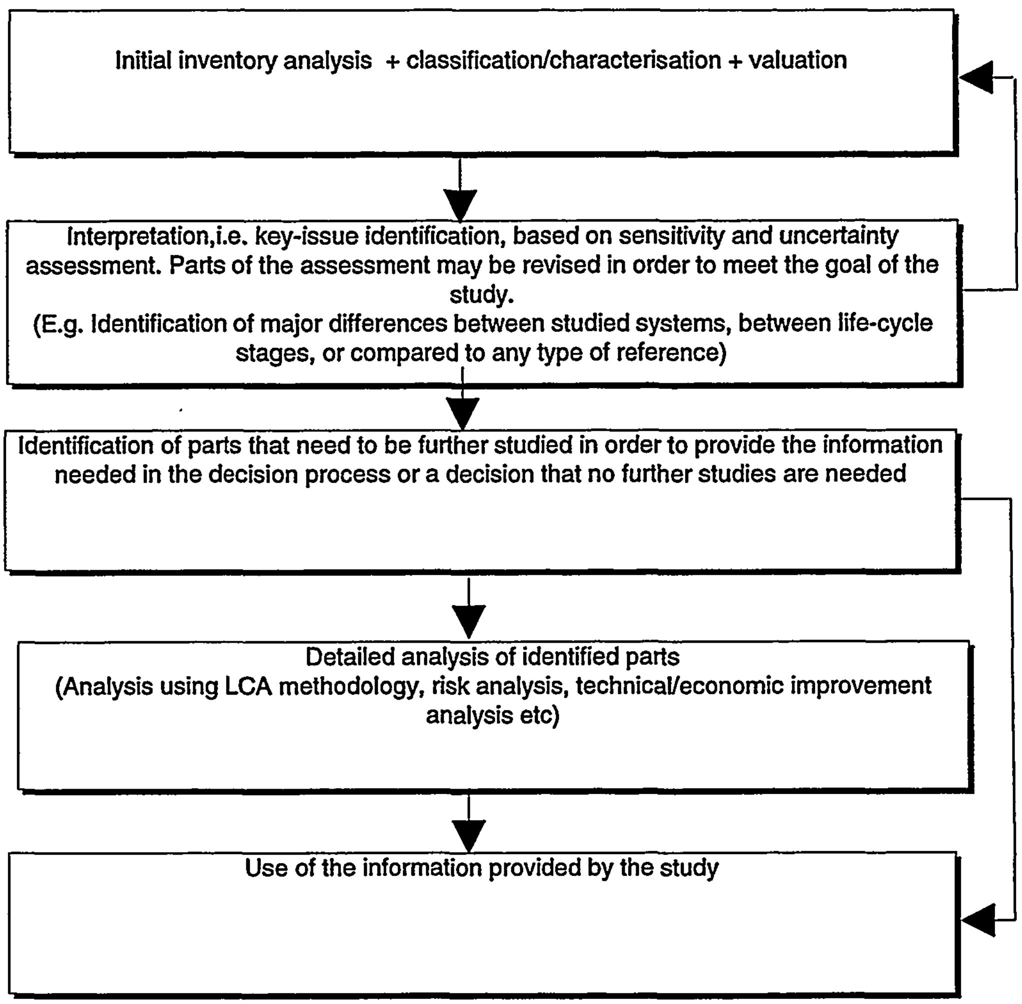

Figure 6.2. The LCA procedure (Lindfors et al, 1995b).

In the LCA procedure an initial LCA including an inventory analysis and an impact assessment is followed by an identification of critical areas (key-issues, hot-spots). Based on this identification, decisions can be made to use the information provided by the study, revise the goal and scope of the study, or go into detailed studies of relevant parts. The detailed study may be an LCA, but may also be other types of studies.

An identification of critical areas can in many cases be made based on the results from the inventory analysis. In other cases, the classification/characterisation may be useful in order to identify important aspects of the life-cycle. However, in order to identify which types of environmental impacts that are of importance, i.e. which impact categories are of importance, a valuation is necessary. Thus, valuation methods are necessary in order to identify some types of critical areas.

When valuation methods are used to identify critical areas, an important requirement is that it actually can identify the hot-spots. Data-gaps are from this perspective decisive. If no valuation weighting factors are available for an impact category, the method is not very useful from this perspective. Another important aspect is that the method should be able to identify areas which are judged to be critical by some of the stakeholders. Thus, interventions and impacts which are controversial should be identified. From this aspect, it may be a good idea to use several complementary valuation methods and data-sets.

7 Final Discussion and Conclusions

There are values involved in the valuation. These are not very often discussed in relation to LCA, an attempt is however made in chapter 2. An important result is that not only the valuation weighting factors, but also the choice of valuation methodology and the choice to use a valuation weighting method at all, are influenced by fundamental ethical and ideological valuations. Since there is no societal consensus on these fundamental values, and never will be in an open democratic society, there is no reason to expect consensus neither on valuation weighting factors, nor on the valuation method or even on the choice of using a valuation weighting method at all. It can therefore be expected that several approaches for valuation will develop including several different weighting sets.

Another result of the discussion on values is that the ethical and ideological valuations are often made implicitly in the choice of method, data etc. Thus, the value-related decisions are only rarely taken explicitly in the development of the methods and the data. As long as the valuations are made implicitly, and almost subconsciously, it will be difficult to discuss the values and the implications of different standpoints.

Currently available valuation methods were briefly discussed in chapter 5. It can be concluded that currently available valuation methods do give different results in actual applications. Another important aspect is that all currently available valuation methods have data gaps limiting their usefulness.

From a critical standpoint it is suggested that straight distance-to-target methods are not valuation methods at all but instead some sort of extended normalisation procedure. This is because straight distance-to-target methods try to avoid the actual weighting by assuming that all environmental problems are equally important. Since this assumption apparently has never been justified, the usefulness of distance-to-target methods as valuation methods can seriously be questioned. Other problems connected with distance-to-target methods include justifying the choice of a specific equation for calculating the weighting factors and defining the targets.

Other currently available valuation methods can be questioned on several grounds, they have major data gaps, some of them are based on more or less controversial characterisation methods, they have not been scientifically evaluated, and some of them will produce strange results either compared to other similar evaluation studies or common opinions in society. It is therefore still an open question whether currently available valuation methods produce useful and reliable information.

The discussion in chapter 6 on different applications concluded that in many applications, the expected result from an LCA is an identification of critical areas rather than a one-dimensional score. This makes the need for valuation weighting methods somewhat more limited compared to what is sometimes suggested. It was also suggested than in many situations, decision-makers will look for Pareto-optimal solutions in which one or several problems is solved while not significantly increasing other problems. In this situation, there is no need for a one-dimensional valuation weighting method.

There are however situations in which there is a need for valuation weighting methods. In some applications in connection with product development, e.g. when a designer is expected to do the LCA him- or herself, there may be a need for one-dimensional scores. An application for valuation methods which may be of large importance is as a part of an LCA procedure for identifying critical areas. An important requirement in this application is that the method actually do identify the key-issues (false negative answers should be avoided, false positives are acceptable).

In chapter 4, different approaches for valuation methods were discussed. When developing valuation weighting methods, which of these should be further pursued? This question is also related to the discussion in chapter 3 on what type of information the valuation should be performed, i.e. where in the cause-effect chain can the border between classification/characterisation and valuation be placed? Below, some pros and cons of the different approaches will be summarised.

As already concluded, straight-distance-to-target methods are of limited value as valuation methods. If such methods are to be further developed, they should be developed as parts of either panel or monetarisation methods.

Monetarisation methods may be used on information both early and late in the cause-effect chain. If it is based on information late in the cause-effect chain, the cost estimates may be called damage costs. It would be interesting to investigate to what extent damage costs could be used as a basis for LCA valuation methods. It is still an open question whether it is possible to calculate damage costs for all relevant types of impacts and interventions. A natural starting point would be to review the work that has already been done by environmental economists and others to see what type of information is available, what type of information could with reasonable efforts be gathered, and what information would be prohibitively difficult to find.

Measures of a society's willingness-to-pay are most often evaluated on information earlier in the cause-effect chain. An advantage of using a society's willingness-to-pay as the basis for the LCA valuation method is that the results would then follow already established societal policies (if the willingness-to-pay measures are a function of these policies). This can in some cases also be a disadvantage, if the valuation method is used to establish a new policy, a circular decision basis could result. However, if the policies based on LCAs are to be based on already, in other areas, established policies, this is not a problem. This is thus a type of valuation method which could be further developed.

In some application areas, notably in connection with product development, it was explicitly noted that the values used should be the values of different stakeholder groups. In order to capture the values of different groups in a society, panel methods are needed (the panel method may also be Contingent Valuation Methods, thus resulting in monetary valuations).

When developing panel methods, there are apparently a number of beginner's mistakes that can be made, especially for people not trained in social and behavioural sciences (like most LCA practitioners including this author). There is thus a need for close co-operation with people with relevant backgrounds. When panel methods are developed, one important question concerns what type of information the valuation is to be based on, results from the inventory analysis, "environmental threat" or "areas for protection" based classification/characterisation, normalised data or not, if so, what type of normalisation data etc. The answer to these questions are related to the cognitive capabilities of the involved people. This in turn is partly empirical questions which can be evaluated.

As noted in chapter 3, the "SETAC-framework" has largely been developed from the wish to separate parts which are based on environmental sciences from parts based on ethical and ideological valuations, primarily focusing on the classification/characterisation subcomponent rather than the valuation. It may thus be the case that the framework is suboptimal if the valuation is taken as the starting point. In the case of a valuation-driven impact assessment, a slightly different framework may be more useful. This framework will however then depend on the chosen valuation method. When deciding on a framework, a certain flexibility is therefore necessary. In the long run, it may prove unpractical to include the requirements of a classification/characterisation step and a valuation-driven impact assessment in the same framework. If so, alternative frameworks, used in parallel may be a practical solution.

In this report a distinction has been made between using valuation methods and drawing conclusions. It will normally always be necessary to draw conclusions from a study. These can however normally not be based only on valuation results. Other aspects, such as data uncertainties, qualitative information, judgements on the importance of data gaps, judgements on the importance of assumptions, etc., must normally also be considered when conclusions are to be drawn. Results from the valuation are thus normally only a part of the background material when conclusions are to be drawn, and in many situations, not even a necessary part.

Acknowledgement

This study was financed by IVL's long-term research programme which in turn was financed by Swedish industries and the Swedish Environmental Protection Agency.

When writing this report, discussions with a lot of people has been very useful. At IVL, especially Lars-Gunnar Lindfors and Viveka Palm has contributed. The discussions within SETAC-Europe's working group on Life Cycle Impact Assessment (chaired by Helias Udo de Haes) and it's subgroup on valuation (chaired by Erwin Lindeijer) has also been a great inspiration for this work. Comments by Reinout Heijungs on a draft are appreciated. Chapter 2 was largely influenced by the report by Stig Wandén (1993).

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