PART 3

SCIENTIFIC BACKGROUND

Reading guidance

This is Part 3 of the set of publications entitled 'Life cycle assessment: An operational guide to the ISO standards'. Part 1, 'LCA in perspective', is a short booklet describing in broad terms the purpose, role, fields of application and limitations of life cycle assessment. It is targeted principally at parties commissioning an LCA and parties using the results of such an analysis. Part 2 consists of two parts: the actual Guide (2a) and an Operational annex (2b). Its target audience is those concerned with actual execution of an LCA. Depending on context and complexity, this may be one person or an entire research team with various backgrounds, such as process technology, product design, abatement technology, ecotoxicology, and so on.
The present Part 3, 'Scientific background', sets out the foundations and arguments for the methodological choices made, the alternative choices that are available, and much more. It is designed to encourage scientific debate on the progress made to date and as a reference book for those wishing to learn more about the rationale behind the rules presented in the Guide.

The presentation of the proposed working method for LCA is structured according to a framework on which there is worldwide consensus and which forms the basis of a series of ISO standards. This framework breaks down the LCA procedure into four distinct phases:
- Goal and scope definition;
- Inventory analysis;
- Impact assessment; and
- Interpretation;

each of which comprises a number of distinct steps. The present Part has a chapter devoted to each of these four phases.

By way of introduction, Chapter 1 discusses the general theoretical foundations and modeling aspects of LCA, considers procedural aspects and outlines the stepwise structure of the present LCA Guide. Chapters 2 to 5 are then devoted to the four phases and constituent steps of an LCA, as specified above, with each step discussed according to a fixed format (as far as is useful):
- topic, providing a brief description of the scope and function of the step;
- developments in the last decade, providing an as comprehensive as possible description and analysis of relevant developments since 1992;
- prospects, describing developments anticipated in the future;
- conclusions, presenting conclusions on the best available practice for the step in question, as recommended in this Guide;
- research recommendations, outlining suggested topics for further research.

Chapter 6 provides a comprehensive bibliography of all the literature sources cited in the preceding chapters.
Appendix A lists all the external contributors to this guide, while Appendices B and C report the results of two desk studies performed as part of the preparatory work for this guide. Appendix B reviews the desk study on the fields of application for LCA, Appendix C that on partitioning economic inputs and outputs to product systems (allocation).

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1. General Introduction

As defined in ISO 14040 (1997E), Life Cycle Assessment is a "compilation and evaluation of the inputs and outputs and the potential environmental impacts of a product system[1] throughout its life cycle". According to ISO 14041 (1998E, p. 2) "A product system is a collection of unit processes connected by flows of intermediate products which perform one or more defined functions. […] The essential property of a product system is characterised by its function, and cannot be defined solely in terms of the final products".[2] In the present Guide the terms 'economic process' or 'economic activity' have been used as synonyms alongside 'unit process' to refer to any kind of process producing an economically valuable material, component or product, or providing an economically valuable service such as transport or waste management. Economic or unit processes (often abbreviated to 'processes') are the smallest portions of a product system for which data are collected during execution of an LCA. They are linked to one another by flows of intermediate products and/or waste for treatment, to other product systems by product flows, and to the environment by elementary flows[3] (ISO 14040, 1997E), i.e. inputs from and outputs to the environment, also referred to as environmental interventions[4]. These environmental interventions may have impacts on the environment by way of environmental processes.

LCA takes as its starting point the function fulfilled by a product system. In principle, it encompasses all the environmental impacts of resource use, land use and emissions associated with all the processes required by this product system to fulfil this function - from resource extraction, through materials production and processing and use of the product during fulfillment of its function, to waste processing of the discarded product.

LCA as defined here deals only with the *environmental* impacts of a product (system), thus ignoring financial, political, social and other factors (e.g. costs, regulatory matters or Third World issues). This does not, of course, imply that these other aspects are less relevant for the overall evaluation of a product, but merely delimits the scope of the present Guide.

The complexity of LCA requires a fixed protocol for performing an LCA study. Such a protocol has been established by the International Standards Organisation, ISO and is generally referred to as the methodological framework. ISO distinguishes four phases of an LCA study (see Figure 1.1):

- Goal and scope definition;
- Inventory analysis;
- Impact assessment;
- Interpretation.

[1] In this Guide, 'product system' is synonymous with 'function system'.

[2] We have endeavoured to ensure that the terminology employed in this Guide is mutually consistent as well as consistently used. Thus, the definitions in this Guide are in basic conformity with the intentions, though not necessarily the letter, of the ISO 1404X series of standards. In a limited number of cases the definitions adopted here deviate from those of ISO, for reasons explained in the text. For the definitions employed in this Guide, see the Glossary.

[3] According to ISO 14040 (1997) 'elementary flow' refers to "(1) material or energy entering the system being studied, which has been drawn from the environment without previous human transformation; (2) material or energy leaving the system being studied, which is discarded into the environment without subsequent human transformation".

[4] In this Guide 'environmental interventions' is the preferred term, this being considered broader than the ISO term 'elementary flow'. From the ISO definition of the latter (see above) it is unclear whether 'elementary flow' also covers land use, while land use is becoming an increasingly important issue under this heading. Environmental interventions thus include both environmental flows and land use.

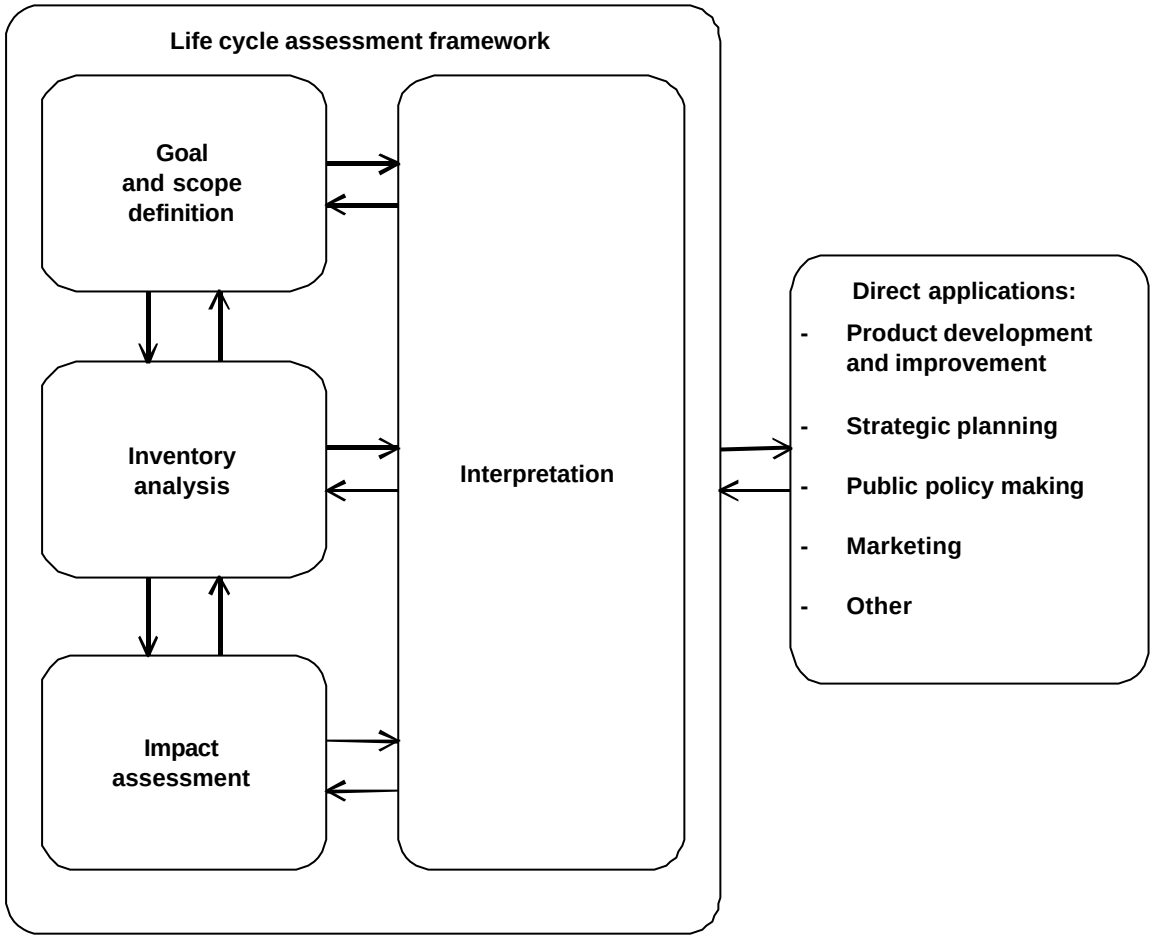


Figure 1.1: Methodological framework of LCA: phases of an LCA (source: ISO 14040, 1997E).

In 1994 ISO established a technical committee (TC 207) charged with standardising a number of environmental management tools, including LCA. To date, four international standards have been published by ISO on the topic of LCA:

- ISO 14040 (1997E): 'Environmental management – Life cycle assessment – Principles and framework' ;
- ISO 14041 (1998E): 'Environmental management – Life cycle assessment – Goal and scope definition and Inventory analysis';
- ISO 14042 (2000E): 'Environmental management – Life cycle assessment – Life cycle Impact assessment';
- ISO 14043 (2000E): 'Environmental management – Life cycle assessment – Life cycle Interpretation'.

These ISO International Standards are important in providing an international reference with respect to principles, framework and terminology for conducting and reporting LCA studies. The ISO standards do not, however, provide a 'cookbook' of step-by-step operational guidelines for conducting an LCA study. Although the ISO Standards contain steps that shall or should be considered when conducting an LCA, these are not ordered in stepwise fashion.

The general aim of the present Guide is to provide just such a stepwise 'cookbook' based on the ISO (International) Standards, as available to the authors at the time of writing (Spring 2000), by operationalising the various elements and requirements of these standards into a 'best available practice' (BAP) for each step. This general aim has been achieved by rewriting the guide of Heijungs *et al.* (1992) to include all the relevant developments - as known to the authors at the time of writing (Summer 2000)[1] - that have taken place since publication of that original Guide, proceeding from the ISO standards and with particular reference to the work undertaken within the SETAC LCA community. The original guide has also been revised to cater for a number of user wishes and requirements, an inventory of which was

[1] This means that a number of more recent developments have not been considered.

undertaken among Dutch users of the LCA Guide immediately prior to start of work on the present Guide (Van Drunen, 1997).

In order to operationalise the LCA methodology some additions to the ISO Standards have been necessary. On some points it was also necessary to deviate from these standards, but only when the rationale for doing so was particularly significant (for example, with respect to the stepwise structure; see below). All these additions and deviations are comprehensively documented and justified in the present Guide.

As far as possible this Guide has been organised in a stepwise manner, reflecting the sequential mode of implementing an LCA study. Although it is recognised that LCA is a highly iterative tool, iteration is not interpreted as a step in its own right as such steps may be required at any point of an LCA study. Hence the Guide has been set up as a 'rational reconstruction' of an actual LCA exercise.

This document consists of three Parts: 1, 2 (a and b) and 3. Part 1 - 'LCA in perspective' - provides a general introduction to LCA and includes a discussion of the possibilities and limitations of LCA and the organisations involved in LCA .

Part 2 - 'Guide' - consists of two parts: a ('Guide') and b ('Operational annex'). Part 2a provides an introduction to the procedural design of an LCA project, and guidelines on the best available practice for each of the steps distinguished in actual performance of an LCA study. Guidelines are provided for two levels of LCA sophistication: simplified and detailed. The latter proceeds from a well-defined set of assumptions or simplifications, which are described in Section 1.2.2.3. The simplified level has been introduced to provide a vehicle for performing faster and cheaper LCAs, which may well be sound enough for certain applications. Thus, the two levels of sophistication relate to different decision situations linked to different methodological choices. On certain points of detail there may often be good reason for undertaking more in-depth analysis than can be provided even by the 'standard' detailed LCA. This kind of in-depth analysis has not been specified here as a separate method but as options for extension. It is important to note that if such options are employed this means a deviation from the assumptions and simplifications described in Section 1.2.2.3 and from the practical points of departure of each LCA phase as described in Part 2a. Both types of decision situation and the methodological choices involved are elaborated in the present Guide. The guidelines for simplified LCA are largely in line with the ISO standards, but not entirely. For example, the allocation procedure recommended for simplified LCA does not comply with the stepwise ISO procedure described in ISO 14041 (1998E). The guidelines given for detailed LCA comply fully with the various ISO Standards mentioned, however, although they are elaborated here at a more operational level. Many of the optional extensions do not fit into the ISO framework. An example is the partial introduction of market mechanisms (lacking in the inventory of detailed LCA), which lead to a type of substitution very different from that described in ISO 14041. In Part 2b the most up-to-date operational models and data associated with the best available practice for these two levels of sophistication are provided as a separate document. This has been done to facilitate updating of these operational elements, most of which are likely to change regularly[1]. Part 2b thus operationalises the guidelines of Part 2a.

Part 3 provides the scientific background to the study as well as a reasoned justification of all the choices made in designing a best available practice for each individual step of the four phases of an LCA.

Below, the general theoretical foundations and modeling aspects of LCA are first discussed. Procedural aspects are then considered and the stepwise structure of the present LCA Guide outlined. Finally, further reading guidance is provided for the remaining chapters of this Part of the Guide.

[1] Whether such updates will actually take place on a regular basis is still uncertain. This would require time and funding and no appropriate arrangements have yet been made.

1.1 General theoretical foundations of LCA

1.1.1 Introduction

The purpose of this chapter is to explore the scientific basis of life cycle assessment. The argumentation is grounded in several disciplines, including philosophy of science, systems analysis, logic and mathematics.

There are various reasons for including a chapter on this subject in the present Guide. Most importantly, it provides the ultimate foundational basis of the entire LCA method as described and developed in this Guide. As such, it can be seen as contributing to the methodology of developing an operational method for LCA. The foundational issue has been raised a number of times in the last few years, especially through the explicit recognition by a number of authors that value judgements are, ought to be, or ought not to be part of the LCA method. In this respect we would mention the 'values debate' around the conception of the ISO standard (Owens, 1997a; Hertwich & Pease, 1998; Marsmann *et al.*, 1999), the incorporation of cultural perspectives in Impact assessment (Hofstetter, 1998; Goedkoop & Spriensma, 1999), the participation of stakeholders when making choices (Tukker, 1998; Bras-Klapwijk, 1999; Lundie, 1999) and, in a quite different direction, attempts to develop inductive approaches to LCA (Heijungs,1998). Several of these developments refer to or build on post-modern (or post-structuralist) movements that have migrated from the humanities to the harder sciences. It should be noted that these developments are not universally welcomed in the humanities or in the social sciences either and that their inclusion in the field comes in many shades, from mere acknowledgement through to radical surrender. In LCA, the debate is not essentially different from that in other fields of study; see, for example, Sokal & Bricmont (1997).

There are other important reasons for including the present chapter. The foundational issue is one that will inevitably be raised whenever there is disagreement among parties: between industry and the environmental movement, for example, between individual industrial sectors or companies, between consumer organisations and government, and so on. While earlier guides to LCA focused on unambiguous and transparent reporting of data, assumptions and methods, it is now clear that this does not suffice. Some differences in data may be caused by differences in the scope, geographical boundaries and so on, of the LCA under review. There are also differences that may be argued to originate from differences in attitude, however. Trip rates of returnable bottles, for instance, can be estimated pessimistically or optimistically, and may therefore differ between technophobes and technocrats. The use of quantitative safety factors for toxic mechanisms that are poorly understood provides another instance where diverging frames of the various parties may lead to disagreement on the findings of an LCA. This brings us back to the topic raised above. At the same time, though, it also points the way to the idea of validating the results of an LCA. Bringing cases to a hypothetical court then lead us to consider a series of crucial questions. The first is: does LCA speak the truth and nothing but the truth? Or does an LCA only provide a certain measure of evidence? How can we ensure that assertions made by or with an LCA have a certain truth content? Can we develop an idea, be it only conceptual, of this content?

A final reason is that an obvious need for such a chapter emerged during the planning and writing of the other parts of the Guide. There are certain topics such as the marginal-average debate, for instance, that exert an influence at so many separate places that dedicated discussion of these topics is a more elegant form of treatment than incomplete references on many different occasions, with excessive repetition. Moreover, many of these recurring themes are closely related. The purposes of LCA, the questions posed in an LCA, the methodological means of achieving answers to these questions, the principles that underlie the modeling of the economy-environment interaction: these are all logically related and deserve coherent treatment.

The structure of the present chapter on foundational issues is as follows. We first discuss the purpose of LCA, in operational terms (Section 1.1.2). As the substance of an LCA is closely related to its purpose, the issues involved are examined in relation to the main types of questions addressed by LCA (Section 1.1.3). In this Guide we focus primarily on the use of LCA for answering structural questions. Next, the methodology of LCA is discussed (Section 1.1.4). This leads us, in subsequent sections, to an extensive treatise on the LCA model developed in this Guide.

1.1.2 The purpose of LCA

By way of an initial reconnaissance of the field considered in this chapter we first discuss the exact purpose of LCA. It should be clear from the outset that what we record here is not the 'one and only' purpose of LCA, but what *we* propose to be the purpose. Some may be disappointed by what they regard as excessively modest aims, while others will hold that LCA will never fulfil the stated purpose. Although we agree that the purpose elaborated here is to be viewed as an ideal that will never be fully achieved, we see it as guiding LCA in a certain direction.

> *The purpose of LCA is to compile and evaluate the environmental consequences of different options for fulfilling a certain function.*

A priori, we thus restrict LCA to environmental consequences. Nonetheless, the exact meaning of this term may still be subject to debate. For instance, inclusion of resource depletion may be controversial, as some see this as an economic problem only. However, we exclude economic consequences, such as cost effects or unemployment, social consequences, such as violation of human rights, and many other types of consequences. We do so not because they are unimportant, but because we wish to focus analysis solely on the environment. As real life decisions are ultimately made on the broader basis, it is important that LCA remain comparable to allies types of analysis in other fields as far as is reasonably possible.

The next item to be noted is that 'purpose' relates to the fulfilment of a function. Thus, plastic and paper are not compared on a per kilogram basis, since this bears no relationship to 'function'. For the same reason, alkyd and acrylic paints are not compared on a per litre basis. Functions are services that are embodied in material objects, often referred to as goods or products[1]. Certain functions involve material products that are intimately related to the potential function they fulfil. For example, the product 'beer' is virtually congruent with the function 'having the pleasure of drinking a beer'. However, it embodies only part of that potential function, because other products (like a glass) may also be required. In the case of other functions the relationship with a (set of) products is rather less direct. For example, the products 'paper', 'pen', 'envelope', 'stamp', 'desk' and 'lamp' are together required for the function 'writing and sending a letter'. In all cases, however, there are products involved that play a very indirect role, in the past, in the future, or in a different part of the world. Beer is produced using ingredients (water, barley, etc.) as well as ancillary goods (electricity, etc.) and also requires capital goods (breweries, etc.). These 'second-order products' are themselves also produced, requiring a third-order level of products. A comparative analysis of the environmental consequences of different methods of beer production should naturally include relevant environmental impacts for at least several orders of these 'upstream' products and production operations. The same applies to the 'downstream' side, which in the example of beer would cover flushing the toilet and cleaning the glass, as well as the operation of the entire waste water treatment system, its construction and its demolition.

We see how naturally the focus on a function opens the door to a comprehensive analysis of the entire life cycle of the function. Observe the phraseology 'life cycle of the function' rather than the more colloquial 'life cycle of the product', to signal the fact that the pivot of the analysis is the function, not the material product that is colloquially associated with it. We also see how naturally the concept of a 'product system', consisting of the several life cycle stages, enters the arena. This will be elaborated further in the subsequent chapters on Goal and scope definition and Inventory analysis.

When speaking of a function, one may indeed have in mind minor functions in the private sphere, like 'writing and sending a letter'. However, there is nothing to prevent us from widening the scope to include broad societal functions, like 'transporting 2 million commuters between the city of Chicago and the suburbs'. Both minor and major functions should in principle be covered by LCA, although the practical methods employed in the analysis may be very different. It is only through an explicit restriction of scope that we can opt for a certain breadth of function as the topic of LCA.

The stated purpose of LCA allows for two quite distinct modes of Interpretation. In the first, the question of interest is the contribution of a particular way of fulfilling a certain function to the entire spectrum of environmental problems as they currently exist or are being created. Using LCA to answer this question is referred to as doing a descriptive LCA, although other terms like retrospective LCA, level 0 LCA, and status quo LCA are also encountered in the literature. In the second mode of interpretation the emphasis is on change. The analysis then addresses the environmental changes resulting from a change from or to a particular way of fulfilling a certain function. This change may assume a variety of forms, illustrated

[1] Products usually comprise both goods and services. Functions may be seen as services.

by ‘drinking one more beer’ and ‘drinking a different brand of beer’. The use of LCA for answering this second type of question is referred to as doing a change-oriented LCA, or sometimes a prospective LCA, or a level 1 (or 2 or 3) LCA. Both types of LCA Interpretation are deemed useful, difficult or state-of-the-art, depending on the perspective of the study in question as well as personal preferences. There is even evidence that while most LCAs are intended to be change-oriented, actual practice is descriptive, or that current practices in fact coincide. Whatever the case, in a chapter treating the foundations of LCA we consider that the two should be kept conceptually separate and discussed as such, even though the later chapters will focus on the change-oriented mode.

1.1.3 The questions addressed by LCA

Our basic point of departure is a decision situation in which a choice is to be made between a number of alternatives. We distinguish between the questions to be answered and the way the answers can be produced. The questions are the starting points, and one or more simple or complex models are the means for answering them. The questions are of the form:

> *What are the environmental[1] effects of choosing option A rather than option B for fulfilling a certain function?*

Restricting ourselves to the Inventory phase, effects are to be considered in terms of environmental interventions[2]. We distinguish three main types of question, related to three main types of decision:[3]

- occasional choices, concerning one-off fulfilment of a function;
- structural choices, concerning a function that is regularly supplied;
- strategic choices, on how to supply a function for a long or even indefinite period of time.

What specific questions are to be answered for each type of choice? Without pretending to be complete, we would consider the following questions relevant for developing a systematic approach to answering them. The category of occasional choices refers to one-off decisions made by individuals that have a negligible influence on society. Concrete examples include:

- mode of travel: should I take the high speed train or the plane to my meeting in Paris next week?
- beverage containers: should I use a china cup or a paper cup in the lunch facility I happen to be visiting today?
- waste management: should I put this particular piece of waste paper in the paper container or the organic wastes container?

The time horizon of the decision and function fulfilment is short. In contrast, the time horizon of effects in the chain may be quite long. In specifying the effect chain, one may take into account simple or more complex mechanisms, simple or complex model forms, one may use one or several models, etc. A current choice does not influence the past, so capital investments like building the aircraft to be used next week play no part in occasional choices. However, the extra passenger may lead to earlier replacement of the aircraft, which then leads to earlier and possibly greater environmental interventions. Similar lines of reasoning can be developed for the other examples.

It should be noted that occasional choices may be frequently repeated, as when a business traveller chooses between train and plane on every business trip. The same applies to the manager of a waste treatment plant making daily decisions on optimum operating conditions, depending on the incoming flow of waste. If a decision can be placed in such a framework, it is no longer an occasional choice.

The category of structural choices refers to choices made by individuals as well as firms relating to decisions that affect society in a limited and readily reversible manner. Examples include:

- mode of travel: should I take the high speed train or the plane to my weekly meetings in Paris?
- beverage containers: should I use a china or paper cup in the lunch facility at my office, every working day?
- waste management: should I put my daily flow of old newspapers in the dustbin or the paper recycling container?

[1] Here, we might add “and other” to allow for coverage of social and economic effects.
[2] Non-environmental effects would fit into the same framework.
[3] This categorisation of questions is close to, but nevertheless differs in some respects from the categories short-term optimisation, mid-term improvements and long-term societal change.

In this case the time horizon of function fulfilment may be long. The context for my decisions in terms of facilities required is given in the short term but the constraints are eased each year, as investments and wear alter the capacities of the various technologies installed. For instance, even a small but continuous flow of extra discarded paper will lead to adjustments in investments at paper mills. The choices illustrated are for consumers, but they may similarly apply to business enterprises, as when a company decides its employees are to make their business trips to Paris by train. A company's decision to give its employees a lease car or a free annual public transport ticket is also in this category. The essential feature is that the choice is reversible, involving no major investments, for instance. This sets limits on the scale of the decision. Thus, if a government were to decide that every citizen be issued with a free public transport ticket, this would require major investments in the railway system, which is to be regarded as irreversible in practice, and we would then categorise such a decision as a 'strategic choice'.

This final category of strategic choices refers to choices by individuals, firms and governments relating to decisions that affect society in a substantial and practically irreversible manner. Examples include:

- mode of travel: should the government invest in high speed railroads or in airports?
- beverage containers: should society as a whole opt for re-usable china luncheon crockery rather than throw-away cups and plates?
- waste management: should cities introduce separation of waste flows into different fractions, with a concomitant set of waste processing and recycling facilities?

The time horizon of the activities involved in function fulfilment is longer for strategic choices than for occasional choices. Structural choices have a similar or more limited time horizon. There may be a delay in the start of the activities in question, as with the investments required in the high speed train example. The time horizon of functioning may be limited to a few years, as with lunchroom kitchens, or may extend to centuries, as with rail infrastructure. Regarding the effects in the chain, one can again opt to consider a shorter or longer period, for example with respect to the time taken for recycling effects to fade off. Again, one or several models may be used, incorporating simple or complex mechanisms and with a simple linear or complex non-linear form. The essential feature is that the choice leads to extensive changes with a high degree of irreversibility, since the investments are so large that it is very unlikely that such a decision will easily be reversed.

It may be difficult to adequately categorise all the choices encountered in practice. Though certain choices may initially appear to be structural, they may be argued to constitute strategic decisions. For instance, if airline companies base their investment plans on current aircraft usage, one extra passenger weekly may lead to more investments. Similarly, an occasional choice for a fluorescent lamp is a decision for 5 years, with possibly even repercussions for power plant investment plans. It is consequently not so important in which category a given concrete choice is placed. What is important, rather, is that the user of LCA is aware of the fact that every category of question highlights certain aspects while ignoring others.

In this Guide we concentrate on the structural choices. That does not imply we consider the occasional and strategic choices uninteresting or beyond the LCA domain. Indeed, we see these as representing useful questions. However, they require a modeling set-up deviating markedly from that needed to support structural choices. We have therefore chosen to leave the elaboration of these categories of choices to other projects. In our opinion the approaches developed by Clift and co-workers and by Weidema and co-workers may be useful for LCAs having occasional choices as their starting point. Strategic choices, on the other hand, call for an approach that would draw more heavily on elements of scenario analysis and partial or general equilibrium modeling.

1.1.4 The methodology of LCA

The terms 'method' and 'methodology' are often used to indicate what essentially is the method. A method is a structured way to achieve a certain goal: to measure the toxicity of a compound, for example, or construct a bridge. A method consists of rules, recipes, formulas, descriptions and so on. Even the Mosaic commandments can be seen as providing a method to live a life in accordance with the principles of God. In any case, a method is not a scientific undertaking. It may be the result of such an undertaking, however, and it may also guide such an undertaking.

Methodology, on the other hand, also referred to as the philosophy of science, is a science that studies means of developing methods that can be labeled scientific. More specifically, epistemology studies the sources of our knowledge. The source of the Mosaic commandments is divine revelation. This source of

knowledge will not be considered scientific; some will deem it superior to scientifically derived knowledge, while others will deem it inferior or simply not believe it. Intuition, tradition, narratives and historical accounts are other sources of knowledge that are not scientific, although that does not mean they are 'false'. Surprisingly perhaps, both approaches (i.e. science-based and not science-based) are possible with LCA. They in fact lead to two different methods for LCA, and correspond to two different Interpretations of the objective of LCA.

On the one hand stands the methodology whereby which the adequacy of LCA theory is determined by comparing theoretical predictions with actual, observed phenomena. Here, LCA is interpreted as part of science, as using the scientific method, be that interpreted as nomological-deductive (Nagel), hypothetical-deductive (Popper) or inductive (Carnap)[1]. The only thing that is accessible for testing empirical relations and therefore able to provide grounds for confirmation or refutation is observed change. If an additional unit of product function is superimposed on a constant background of economic activity and environmental pollution, the change in observed activity and pollution can be argued to be due to the additional unit of function. In practice, autonomous changes in economic activity and pollution may be so large that we cannot speak of a constant background. We therefore have a violation of the *ceteris paribus* assumption. In many fields of science, like biology and sociology, however, this is a quite common state of affairs. A frequently adopted approach is then to analyse the complicated structure in question by way of a number of simpler steps. These steps then correspond to portions of accepted models, disciplines or causal relationships. Relevant examples in the context of LCA are process technology, micro-economics, multi-media fate models, dose-response functions and so on. The important point to bear in mind is that these previously established 'building blocks' are assembled as a means of predicting the changes in economic activity and environmental pollution due to an additional unit of product function. In other words, the approach is suitable as an epistemological basis for the change-oriented variant of LCA. For the normative element of LCA, the tools are not those of science but are in a similar vein, including multi-criteria analysis, multi-objective decision-making and, more generally, decision theory.

On the other hand stands the methodology for dealing with descriptive LCA. This is not amenable to the scientific method (nomological deduction, hypothetical deduction or induction). There is no possibility, not even in theory, of testing empirically the contribution to economic activity and environmental pollution of a unit amount of product function. We can neither confirm nor refute statements based on a descriptive LCA. However, some of these statements will certainly be deemed more logical than others. It may be possible to design a set of rules for developing a method for doing descriptive LCA studies which is internally consistent and more or less in accordance with a number of intuitive ideas. The rules forming the basis of such a method are the axioms (postulates, principles) and the method developed from it follows by deduction, employing theorems (propositions) requiring rigorous proof. Given a consistent set of axioms, a method may be classified as false or true in a logical sense. It may be that certain propositions are undecidable, that is, their truth or falsehood cannot be established by means of the axioms. However, a different but equally consistent set of axioms may well lead to a different method, producing different but equally true LCA results.

In applying these two methodologies there may be a substantial overlap of activities. When models are constructed according to the empirical-scientific method, there will be an array of central theoretical concepts. These may be interpreted as hypotheses, which can be refuted, or may be interpreted as axioms, which may or may not be accepted for certain reasons. Statements derived from these central concepts or axioms are true in the empirical science sense if the central concepts have been corroborated, in the axiom sense if they seem 'right'. An example is the 100% rule in allocation, which may be interpreted as a reasonable or just axiom, or (in a specific model structure) as an empirical rule based on the conservation of mass and energy.

1.2 Modeling aspects of LCA

[1] The reader should bear in mind that the term 'deductive' in the realm of logic, for deriving conclusions from premises, embodies a different approach (see below) and that the term 'complete induction' is likewise used in a different sense in mathematics.

1.2.1 Introduction

In the previous section we have seen that there are two fundamentally different methodologies for developing methods for LCA. These two indeed lead to a number of methods within each of the two modes, descriptive and change-oriented. For the descriptive mode of LCA this was already concluded when discussing the variety of options for choosing a set of axioms. However, this open character of the method also holds true for the change-oriented mode of LCA. Although there is one ultimate benchmark for testing the predictions of change-oriented LCA, it is clear that this benchmark is useless in practice owing to the complex nature of autonomous developments in society, the economy and the environment. A plethora of unforeseen events mean that a *ceteris paribus* assumption will never hold for the full LCA model. Even in the absence of such extreme events as economic depressions, volcano eruptions or civil wars, technological change, economic growth, cultural development of tastes, shifts in taxing structure, changing infrastructure and so on are so ubiquitous that the consequences of decisions we wish to observe are mere ripples in a stormy pond. In practice, therefore, comparison of LCA predictions with reality is unattainable. In addition, when a choice is to be made between several alternatives for a given function system, it might only be the effects of the adopted course of action that are investigated. As the effects of the rejected alternatives will never materialise, the grounds for the choice will be based on modeling assumptions, even though the consequences of implementing one alternative might be analysed empirically.

Not only is society *in toto* too complex to analyse and in too great a flux to assume it in any way constant. It must also be acknowledged that predicting the environmental consequences of a simple unit change in the economy is an undertaking of tremendous complexity. The pathways in the economy and environment are long, interwoven and full of time lags and non-linearities. Every prediction will be based on a model, which simultaneously embodies incorporated knowledge and deliberate ignorance. In other words, the predictions made with a change-oriented LCA are based on model calculations, and a model is a simplified representation of real mechanisms and phenomena. The choice of where to introduce which simplification is partly subjective. Different researchers may wish to use different models, corresponding to different assumptions and different simplifications. These researchers will therefore produce different answers to the same change-oriented question. Since there is only a platonic view of the benchmark of reality, it is hard to judge these different answers on the basis of their truth content. The best solution here is to state as explicitly as possible the assumptions and simplifications introduced in modeling the environmental consequences of a given change. Modeling choices are not entirely subjective, however. Depending on the questions asked, some models may be more appropriate than others. If one is interested in short-term effects of a given choice, short-term relations should play a role, including changes in capacity usage of given installations in the model. If the interest is in long-term consequences, the model should indicate consequences of changes in installed capacities. If both short-term and long-term consequences are to be examined, a dynamic model predicting effects in time is the most appropriate.

As change-oriented LCA constitutes the principal focus of the remainder of this Guide, the rest of this section will be concerned mainly with an explicit treatment of our arguments for constructing a particular LCA model and a consideration of theoretical and practical modeling aspects. The theoretical part considers such issues as the general structure of the model and the main directions in which simplifications can be introduced, while the practical part is devoted to several pertinent problems in LCA: potential and actual impacts, non-linearities, spatial differentiation, average and marginal data and software.

1.2.2 Theoretical modeling aspects

1.2.2.1 The reality behind the model

It is useful to gain a clearer idea of what kind of reality is supposed to be reflected by the LCA model. For the sake of simplification, we shall restrict our discussion to the Inventory analysis; an extension to Impact assessment is, in principle, straightforward.

When studying any change in environmental interventions it is necessary to specify:

- a time pattern, e.g. distinguishing between short-term changes and changes in the longer term;

– a reference situation (or more precisely: time path) in which the changes induced by the alternative investigated do not take place, that is the autonomous pattern or a reference alternative.

Figure 1.2.2.1.1 illustrates what is involved in a change-oriented analysis. We see a time pattern of emissions.[1] There is an autonomous pattern (e_0), governed by a complex pattern of economic cycles, environmental regulation, population growth, cultural influences, natural disasters, civil wars and so forth. At a certain moment in time (t_0) an economic actor has the power to decide to implement a choice, which is to be supported by an environmental analysis. This may be a producer redesigning his production process, a consumer buying a certain product, an authority granting a permit, and so on. From t_0 onwards Figure 1 shows two lines: one with some alternative being implemented (e_2) and one with the reference alternative (e_1). The analysis is thus between two parallel systems; it is not a before-after comparison but a with-without comparison.

It is assumed here that there is an immediate effect at t_0, and that there is a complicated difference between the two lines at later stages. This complicated difference is due to upstream and downstream consequences that are introduced with different time-lags. Note that upstream processes occur only after the decision has been implemented.

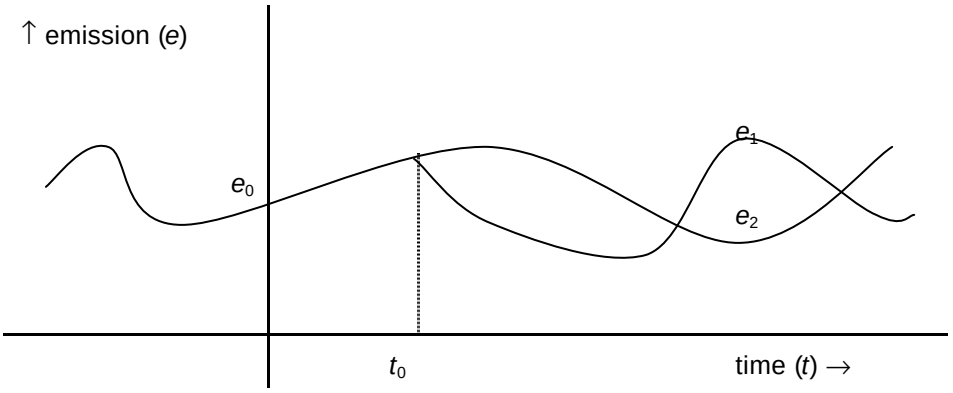


Figure 1.2.2.1.1: Time pattern of an emission before a choice (e_0, to the left of t_0), following non-implementation of the choice (e_1, to the right of t_0) and following implementation (e_2, also to the right of t_0).

An example may clarify this further. A consumer who decides to drink one extra cup of coffee, buying it from a coffee machine, first causes an increase in electricity and water demand. With a delay of a few hours, he will use the toilet, again using electricity and water, but now also involving the waste-water treatment plant. With a delay of a few days, filling of the coffee machine will take place, with some extra filling of coffee (and perhaps sugar and milk) and the cup reservoir required. The same applies to the disposed cup. After days to weeks, the signal of extra coffee consumption will have reached the producers of coffee, milk, sugar and cups, and the waste treatment facilities. These will presumably produce extra coffee, milk, etc. to compensate for the extra use. Next they may also be induced by this and similar signals to adjust their longer-term production and/or investment plans. This latter, indirect effect may also reach the producers of electricity and water, the coffee machine maintenance firm and the firm that empties the waste bins. For instance, it may be the case that an extra waste bin is already being delivered a day later, from stock. We may even extend this example, with bizarre consequences, analogous to the image used to illustrate chaos theory: the butterfly in Peking which causes a storm in New York. For instance, my personal coffee consumption may cause imports to exceed the threshold for constructing a new freight airport. Nobody knows at what levels chaos and instability occur.

[1] For explanatory purposes, this is a one-dimensional conception, representing emissions of CO_2, for example, or damage to the ozone layer or an aggregated weighting result. The focus of the present discussion on Inventory analysis means that emissions will be the typical entity considered, hence the symbol *e*. The generalisation to a multi-dimensional set of environmental consequences, including resource use, is straightforward.

Similar examples may be conceived for any choice: product design, packaging material, mode of transport, etc. There may be all kinds of primary, secondary, etc. effects invoked at quite different moments of time. A general formula for expressing the effects of making a certain choice can be expressed as:

$$\Delta(t_1) = e_2(t_1) - e_1(t_1) \qquad (1.2.2.1.1)$$

where t_1 is the point in time ($>t_0$) at which the difference between the two emission levels is evaluated. ?(t_1) represents this difference, and can hence be interpreted as the difference between the emission level with and without the choice being implemented. In principle, two future states are predicted, one for the reference situation and one following the choice being made and implemented.

In the expression above, t_1 denotes the point in time at which the emission levels are predicted and the difference between them is evaluated. An obvious question is what should be chosen for t_1. A point in time is an infinitely short lapse of time. It may be that one of the processes of the life cycle is actually operating at that point in time, but most of the processes will take place at different moments in time. Therefore, opting for one point in time will discard many emissions from the life cycle, and is therefore not compatible with the principles outlined in the section on the purpose of LCA. It is natural to take a broader time span, for instance taking into account all substances released during the first day or month or decade after the moment of decision. If we wish to cover instantaneous effects as well as effects on the time scale of hours, days and years, we must switch from one moment in time to either the modeling and analysis of time series, or to an integration over an interval of time. In this section we shall discuss the first option, while the second option will be considered as one form of simplification and is therefore discussed later, in the section on model simplifications.

Modeling time series comes down to predicting $\Delta(t)$ for a (possibly large) number of intervals for t. To give a concrete example, we could model the emission of CO_2 during the first week following the decision, during the second week, during the third week, and so on. Or we could do so on a annual or daily basis. Within each time interval, the emissions would be integrated over that time interval. If required, time intervals might be unequal, for instance the first hour, the next week, the next year, and so on. Within an economic-environmental analysis, a quite detailed level of distinct time intervals is readily defensible. The fate and effects of volatile compounds may differ between morning, noon, evening and night. And aircraft noise is clearly perceived differently during day and night time. For certain chemicals, a seasonal distinction between summer and winter seems more appropriate. Hence, while modeling time series may appear straightforward from a theoretical perspective, in practice it leads to massive data requirements, an immensely complex model and a tremendous quantity of results.

The discussion on the temporal domain can be repeated for the spatial domain. In Figure 1.2.2.1.1 there is one independent variable: time. However, we may add three more mode-independent variables, corresponding to geographical longitude, latitude and elevation above ground level. This also means that we now express an emission as $e(x, y, z)$ rather than $e(t)$. There are again three principally different options: to restrict the analysis to a single location, to model spatial differentiation within selected regions and to integrate over the entire spatial domain. The first option is of definite interest for local policy-making, but again seems incompatible with the life cycle concept of wishing to include upstream and downstream effects, which almost by definition take place somewhere else on the earth. In the second option, a number of regions are distinguished for specifying where emissions take place. Again, from an economic-environmental point of view, this may be very relevant indeed. Emissions of acidifying compounds in Scandinavia lead to quite different effects from similar emissions in the Sahara. For volatile organic solvents, a distinction between indoor and outdoor emissions seems appropriate (Potting & Blok, 1994). And for SO_2 there is a distinct difference between urban, rural and marine areas. As with time series, fine-meshed spatial series increase environmental relevance, but lead to spiralling data requirements, ever more complex models and extremely extensive analysis results.

1.2.2.2 The need for a simple model

A model is a simplified representation of part of reality. In the context of LCA, two basic modeling issues are of particular importance.

- LCA deals with complex, interwoven networks of industrial, agricultural, household and waste management activities dispersed over many locations and potentially spanning many decades. The mechanisms governing the dynamics of these activities are of a technical, economic, social, cultural and political nature. The mathematical relationships that describe these real mechanisms are, by principle, non-linear and dynamic and will often exhibit hysteresis and irreversibility. No such a

model of 'true reality' exists, and an LCA model must inevitably introduce a multiplicity of crude simplifications.

– LCA concentrates on the function system for a particular product life cycle, although it is well known that economy and technology are such that any two products are at some stage connected through a common process. The LCA model should therefore be capable of dissecting a product life cycle from this interconnected complex. In particular, this procedure for isolating an individual product life cycle requires the setting of system boundaries and an appropriate allocation procedure. Another option is to incorporate 'all other processes' in a more aggregated fashion, using input-output tables, for example (Lave *et al.*, 1995).

Ideally, one would like to see the environmental interventions (and/or impacts) associated with a particular choice specified in terms of both location and time. For theoretical reasons this is extremely difficult, if not impossible, however. LCA practitioners are therefore happy to overcome the modeling difficulties while maintaining reasonable accuracy for the sum total of environmental interventions, integrated over all locations and infinite time in an assumed steady state. To date, little attention has been paid to aspects of spatial differentiation or specification of dynamic patterns in time, of the form 'emission in Dublin on June 16th'. This has meant that LCA is sometimes said to exclude *a priori* specifications of space and time, and that spatially differentiated LCA or dynamic LCA is almost a contradiction in terms. In this Guide we take a more liberal stance, regarding spatial and temporal integration as two possible steps in the inevitable process of constructing a workable LCA model.

Several other simplifications have also been introduced. Some of these relate to model structure (e.g. linearisation of process characteristics and aggregation of individual companies along sectoral lines), others to mechanisms (e.g. ignoring demand elasticities or changes in use patterns by consumers). Again, we stress that although these simplifying steps facilitate the modeling exercise and may be a normal feature of LCA practice, the explicit incorporation of, say, non-linear relationships and economic mechanisms by no means contradicts the aim or principles of LCA. Hence, where this Guide proposes certain simplifications to the LCA model, this does not preclude additional efforts being made to develop more sophisticated models. This will hold particularly for strategic decisions having large-scale implications, and in such cases dynamic, non-linear, complex modeling may be imperative. This issue will not be considered in the present Guide.

1.2.2.3 Main model simplifications

Although the ultimate aim is obviously to develop a high-quality model having adequate discriminatory power and yielding a consistently valid and reliable outcome, in practice this aim is constrained by a shortage of data, theoretical expertise and the capacity to handle complexity. Inevitably, simplifications must be introduced. While omission of economic mechanisms and spatial detail makes for great simplification, however, it reduces the quality of the analysis results. When making predictions about the functioning of economic processes and their interrelationships, it is obviously a substantial loss to disregard economic mechanisms. The same holds for a wide variety of environmental effects that may be highly dependent on the location of the activities in question. Below, we propose several simplifications to the life cycle Inventory analysis model for use in supporting structural decisions. This exposition should be regarded as providing reference standards for two levels of LCA complexity suitable for different types of decision situation. We first provide a reference standard for a detailed type of LCA, which can be seen as a stepping stone towards an ideal generic method. As already mentioned, deviations from these simplifications are perfectly legitimate if such a need is felt. Such deviations may indeed lead to major improvements in model quality. However, a more simplified model may be perfectly legitimate in situations where the key differences between alternatives are very clear or where a rough indication is a good enough starting point. Alongside the detailed LCA we thus also specify, in Chapter 2 below, a reference for a more simplified version of LCA. Such 'best available practice' types of LCA are 'best' only in the given historic context and the databases and software that history has spawned.

Two of the main directions for simplification follow directly from the discussion on the temporal and spatial detail of the analysis.

The alternative to modeling time series is to integrate over the entire time horizon. Technically speaking this is not in itself a simplification, as the time series must first be modeled before integration is possible. In practice, however, integration opens the road to further simplification. If linear models are used, for example, it becomes irrelevant when exactly in time specific activities occur. In integration, the

basic quantity of interest shifts from the difference between two emission levels at distinct points in time to the time-integrated difference between these two emission levels. Mathematically:

$$\Lambda_t(t_2,t_3)=\int_{t_2}^{t_3}\left(e_2(t)-e_1(t)\right)\mathrm{d}t \qquad (1.2.2.3.1)$$

where t_2 and t_3 delimit the time period covered by the analysis. $?_t$ represents this time-integrated difference. Observe that ? in Equation (1.2.2.1.1) and interrelationships and $?_t$ in Equation (1.2.2.3.1) have different dimensions. If the emission *e* is measured in kg, for instance, ? will likewise be expressed in kg, while $?_t$ will have the unit kg×yr. It seems natural to opt for full time integration when dealing with consequences occurring throughout the life cycle. In other words, taking $t_2=t_0$ and $t_3=\infty$ as integration boundaries seems to be optimal in relation to the stated purpose of LCA. However, it is sometimes advocated to restrict the time horizon from infinity to 100 years, for example. The main arguments given are data availability, policy relevance and uncertainties with respect to the very distant future. Especially in discussions regarding the impacts of landfill and the toxicity of heavy metals and other persistent chemicals this issue is sometimes raised. An alternative here is to retain an infinitely long time horizon but to discount the future by means of a temporal discount rate that gives less weight to events occurring, at least in the model, in the distant future. An operational formula here is:

$$\Lambda_t(t_2,t_3)=\int_{t_2}^{t_3}\left(e_2(t)-e_1(t)\right)e^{-rt}\,\mathrm{d}t \qquad (1.2.2.3.2)$$

where *r* is the temporal discount rate. Typical values of *r* range from 0 to 5%. A discount rate of 5% implies that the importance of effects is halved every 14 years, which means that effects occurring over 100 years hence are virtually ignored ($1/2^{(100/14)} = 0.007$). In Equation (1) the implicit value judgement is that effects now and in the future are valued the same. In Equation (2) the same can be achieved explicitly by setting $r = 0$.

The compromise between taking a single point in time and modeling time series was found to be integration over the entire period in question. We can carry this compromise over to the spatial context. A complete integration of life-cycle-related emissions over the entire globe provides a workable approach that is compatible with the idea of following a life cycle. Mathematically:

$$\Lambda_s=\iiint_{\text{world}}\left(e_2(x,y,z)-e_1(x,y,z)\right)\mathrm{d}x\mathrm{d}y\mathrm{d}z \qquad (1.2.2.3.3)$$

The threefold integration over the three spatial dimensions means that a possible unit for $?_s$ is kg×m^3. In the integration, we have ignored the possibility of integrating only partially over space to exclude, say, Antarctica, the oceans or countries that are not signatories to certain environmental protocols. This option is technically analogous to setting the time horizon to 100 years instead of infinity. We also have ignored the possibility of 'space discounting', because we do not consider it a meaningful option in current LCA. The implicit value judgement made in the spatial integration step is that similar effects in different locations are valued the same in the ultimate evaluation of impacts.

Combination of the compromises with respect to temporal and spatial integration yields the formula:

$$\Lambda_{s,t}=\int_{t_0}^{\infty}\iiint_{\text{world}}\left(e_2(x,y,z,t)-e_1(x,y,z,t)\right)\mathrm{d}x\mathrm{d}y\mathrm{d}z\mathrm{d}t \qquad (1.2.2.3.4)$$

Of course, this formula is of little practical use and we shall not employ it in the remainder of the text. However, it serves to emphasise the basic idea that full temporal and spatial integration of emissions of pollutants and extractions of resources is a principle that is in good agreement with the stated purpose of LCA.

Continuing the discussions on model simplifications, the following choices have been made in the baseline LCA model presented in this Guide:

– Near-complete omission of spatial detail, by not distinguishing between emissions near different kinds of ecosystem, for example. Note that this does not mean that the distances between unit processes is set at zero: transport is merely taken into account. Neither does it mean that all unit

processes are assumed to operate according to the technological state of the art that is representative for a given region. We can still distinguish between the emission characteristics of electricity production at different locations, but merely refrain from specifying where the emission occurs. The only default spatial details that are retained are those specified by a short list of environmental media: air, surface water, soil, sea and sediment.

- Complete omission of temporal detail. Among other things, this means that emissions are specified as total (infinite) time-integrated emissions. This does not mean that operations like storage[1] are left out. Neither does it mean that all unit processes are assumed to operate according to the technological state of the art at a given point in time. We can still distinguish between the emission characteristics of electricity production needed for constructing factory buildings now and that for recycling them 50 years hence. In fact, we recommend doing so.
- Complete omission of non-linearities.[2] This means, for example, that if the production of 1 kg steel is associated with an emission of 5 kg of a substance, the production of 2 kg steel is assumed to be associated with an emission of 10 kg of that substance (hence, fixed input-output coefficients).
- Omission or extreme simplification of most economic, socio-cultural and technological mechanisms influencing operation of the processes considered in the Inventory analysis. For example, the most common economic response to a rise in demand for a product brought about by adopting a certain option for supply of a functional unit is a rise in price and certain buyers of that product leaving the market. This market mechanism is virtually ignored. Income effects are likewise ignored. For instance, if product A is more expensive than product B, switching to B will mean the consumer has more money to spend on other (polluting) activities. This shift is not generally taken into account.[3] The same holds for the phenomenon of people leaving more lamps on if light bulbs are energy-efficient and hence less costly to burn. There are numerous examples of such forms of simplifications. Certain mechanisms are simply ignored (like the two above), while others are taken into limited account. An example in the latter category is given by economic substitution of certain materials by coproduced materials, where a sophisticated economic model would apply cross-elasticities, while some LCA analysts assume full, or zero, substitution, or use this substitution as an artificial procedure for solving the multifunctionality problem, as a kind of allocation procedure. As a baseline, we here ignore all behavioural mechanisms except those related to changes in volume and (see above) simplify the latter mechanisms to a linear approximation. This has, *Inter alia*, important consequences for allocation; see Section 3.9.

[1] To continue the analogy with space: transport is the process that carries an item from location A to location B, while storage is the process that 'carries' an item from day A to day B. Both transport and storage are economic processes which may need fuel, electricity, cleaning and so on, which may emit pollutants and which sometimes lead to a change in the quality (or even quantity) of the product concerned, for either better (wine) or worse (fresh flowers).

[2] In this context, neglecting non-linearities means that the model structure is linear, in the sense of twice as much product requiring twice the amount of materials, leading to twice the emissions and to twice the environmental impact. It may still be the case that the simple coefficients used in the model are derived from highly non-linear models, as with global warming potential, for example.

[3] Unless, for example, the functional unit is expressed in monetary units.

Figure 1.2.2.3.1 shows the model that is assumed to reflect the reality depicted in Figure 1.2.2.1.1 under some of the simplifications discussed above.

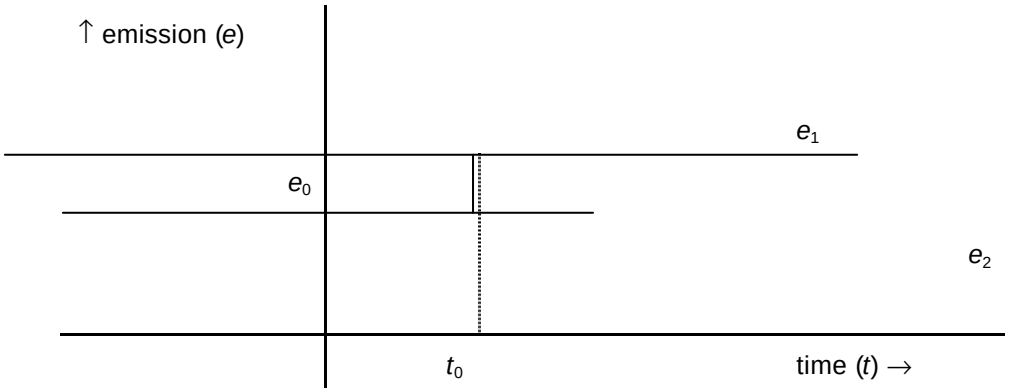


Figure 1.2.2.3.1: Simplified time pattern of the situation of Figure 1.2.2.1.1.

Note that some of the complexities of economy-environment relationships are maintained. For instance, there is usually no aggregation of pollutants into substance groups (like 'heavy metals' or 'organic compounds'). Similarly, economic activities are analysed at the micro-level, with 'steel rolling' or 'paper bleaching' being treated as separate activities. It is, of course, perfectly permissible to introduce further simplifications along these dimensions. They facilitate computation, but at the price of a less relevant answer. In our opinion the baseline simplifications proposed in this Guide represent a good compromise between quality and feasibility for structural, change-oriented LCAs.

1.2.3 Practical modeling aspects

1.2.3.1 Potential versus actual impacts

There is a long tradition of debate on what types of impacts are accessible and/or predictable in LCA, and in particular in life cycle Impact assessment (LCIA). Emissions of hazardous chemicals take place at many different locations and at many different times. The default simplification principle of full space and time integration means that there is no information on spatial and temporal detail in the model. One just specifies the total life cycle loadings, in the form of aggregated releases: 5 mg mercury to water, 12 kg sulphur dioxide to air, and so on. Hence, any attempt to interpret the contributions of these substances to environmental impact categories, such as ecotoxicity and acidification, can only be made without incorporation of spatial and temporal information, even if in some instances such details are available. Examples of items that may differ significantly between locations and times are background concentrations, presence of vulnerable ecosystems, environmental properties like soil composition, temperature and precipitation characteristics, population density and consumption patterns, and many more. Absence of such knowledge, or the lack of scope for including such knowledge, means that the impacts of releases can be estimated in very generic terms only.

As an example, consider the following. If, under average conditions, substance *x* is 20 times more persistent and 5 times more toxic than substance *y*, a release of one unit of substance *x* is considered to be equally hazardous to a release of 100 units of substance *y*. This rule is also applied in non-average situations. In fact, the question whether the situation is average or non-average, and any deviation from the average, is entirely ignored in assessing potential impacts. Examples of questions that are ignored are:

- are background concentrations below or above supposed threshold levels?
- do the releases take place in a densely or sparsely populated area?
- do the releases take place in a vulnerable or invulnerable ecosystem?
- is the fish in this lake actually used for consumption or not?
- is the prevailing wind direction around this chimney towards or away from the sea?

Assessment in terms of potential impacts is sometimes understood to imply a worst-case analysis. This is certainly not an accurate description of the main idea behind 'generic conditions'. However, in a certain sense it points in the right direction, as it is the capacity for causing harmful effects that forms the basis of the assessment, and not so much the extent to which this capacity becomes effective. For instance, one aspect of pollution abatement is to release pollutants at the right place and the right time. Focusing on potential impacts means that such measures are ignored. Even if some further spatial detail were to be added to the models, on top of the seven media already considered, differentiating between densely and sparsely populated areas, for example, the 'potential' nature of the impact analysis would remain largely unchanged, as it still does not take all relevant characteristics into account.
Actual impacts are sometimes contrasted to potential impacts, and their exact nature is perhaps even more confusing.[1] If we move away from full space and time integration to add more and more details with respect to the spatial and temporal characteristics of release and receiving environment, we might say that we are moving into the area of actual impacts. There is a huge difference between full integration and full differentiation, however, and there is hence a broad continuum between potential and actual impacts. Understood in this sense, potential and actual impacts are not so much a pair of opposites. Rather, one may speak of a degree of actuality that is taken into account by the Impact assessment method. It is clear that the amount of information required for assessing releases will grow as the required degree of actuality rises. For assessment in entirely potential terms, it suffices to know a handful of persistency and toxicity characteristics along with some overall parameters like average temperature, average soil pH, average intake of drinking water, and so on. For an assessment in more actual terms, more parameters are needed: temperature of the release region, population density of the receiving region, and so on. An assessment in extremely actual terms is an ideal to be approached but never reached. It would include questions like: what is the body weight of the person walking over there at the moment of release? Is she pregnant? Is she a smoker or non-smoker? An assessment of impacts in extremely actual terms is very complicated, and it is doubtful whether there is any sense to it at all, even outside the realm of LCA. The inclusion of certain aspects of actuality is certainly important in tools like Risk Assessment (RA) and Environmental Impact Assessment (EIA), which are geared towards operating permits and other location-related decisions. For a tool that has comprehensiveness as its prime characteristic, in a first approximation it seems natural for LCA to ignore actuality-determining factors[2]. There is no fundamental reason for making LCA so abstract and general, merely reasons of practicality. Our objective is to construct an operational form of LCA, which implies a simple model. As better types of model and data sets become available in due course, with results amenable to interpretation, the reference method for the detailed LCA model, as well as for the simplified LCA model, can be improved. If a deeper analysis is required, and if the resources to do so are available, there is no reason to stick to detailed LCA. More extended options might be examined and in many situations non-LCA types of analysis may also be relevant. On the other hand, a detailed LCA may be more demanding than is necessary for some more simple types of decisions.

1.2.3.2 Beyond the default simplifications

As stated above, the proposed simplifications provide a crude reference model structure that can always be improved in a wide variety of respects as circumstances dictate. The aim of this section is to discuss some of the ideas that may be introduced for improving on the reference model quality, as specified for detailed LCA. There is one particular direction for further model detailing that has received considerable attention in the literature: spatial differentiation. This is acknowledged by a separate section being devoted to this topic. The other directions in which attempts to escape from the simplifications have been made are described in this section.

So far, little attention has been paid in the literature to temporal differentiation in LCA. Owens (1996, 1997a and 1997b) criticises the lack of "spatial and temporal considerations" in LCA. He mentions rates of emissions, duration and frequency of exposure, and seasonal influences of temperature and sunlight as missing time characteristics. He gives no suggestions, however, as to how these characteristics

[1] The juxtaposition of 'actual' and 'potential' impacts remains ambiguous. Potential impacts sometimes denote modeled impacts, and are thus opposed to measured impacts. Sometimes the term is used to indicate the impact of below-threshold emissions, which, while not an impact in the sense of actual damage to ecosystems, might be described as occupying environmental utilisation space.
[2] See appendix B for a more comprehensive discussion of the relationship between tools and applications.

might be included operationally. Pleijel *et al.* (1999) studied the influences of climate, source strength and time of day of NOx and VOC releases on tropospheric ozone formation in the context of LCA. They found considerable variation. If and how this can be included in general LCA methodology is a question yet to be answered. Finally, Potting (2000) very briefly raises the same issue in a thesis otherwise devoted to the issue of spatial differentiation.

With respect to the inclusion of non-linearities, there is some literature relating to operations research and equilibrium models. Kandelaars (1998) discusses the materials-product chain, a concept that more or less coincides with LCA. The technical elaboration is quite different, however. Some unit processes are described within a framework of micro-economic production functions. Their concaveness contrasts with the linear, homogeneous, Leontief-type model that lies at the heart of LCA. It follows that solutions to the inventory problem cannot be found by normal matrix inversion (see Section 3.10), but that an optimisation algorithm is needed. Among other things this requires a quantity to be optimised, which is generally of a financial nature (e.g. cost minimisation or profit maximisation). Kandelaars shows that such models are feasible. However, all the examples she elaborates are very small product systems, comprising no more than 20 unit processes. It follows that the realism gained by including non-linearities is accompanied by a loss of realism due to a fairly drastic cut-off of ignored flows. It is difficult to say how gains and losses balance out in this and other cases. It remains the normal practice of LCA to place emphasis on completeness rather than elaborateness of mechanisms. Similar cases are provided by Gelderman (1999) and Freire *et al.* (2000), among others.

The issue of including socio-economic, technological and other mechanisms has been discussed mainly in relation to the choices of product alternatives and functional unit and in relation to allocation based on the substitution method. However, the aforementioned use of concave production functions with an economic optimisation objective is obviously also an example of how realism vis-à-vis economic mechanisms can be improved.

Related to the goal definition is the question of which product alternatives are to be deemed comparable and what amounts of these product alternatives are assumed to function as substitutes. Fluorescent lamps are not always appropriate substitutes for incandescent lamps and when they are it is not always 100 hours that substitute for 100 hours, because people tend to keep fluorescent lamps switched on when they leave the room for a while. A strong plea for incorporating real behavioural mechanisms in such situations is made by Wegener Sleeswijk *et al.* (1995) and by the *Groupe des Sages* (Udo de Haes *et al.*, 1996). In concrete cases, empirical usage data on the various product alternatives have sometimes been incorporated (Potting *et al.*, 1995). See Section 2.4 for a more extensive discussion of this topic.
With respect to the allocation issue (see Section 3.9), it should be emphasised first that the substitution method implies incorporation of socio-economic mechanisms in an analysis that at most points ignores such mechanisms. The assumption of the substitution method is that if the product system under study also delivers a coproduct this will obviate the need to produce a product with more or less identical features. In its crudest form, the substitution method implicitly assumes that supply of the coproduct is fully inelastic and supply of the substituted product fully elastic. In general, neither of these assumptions will hold. To reflect relevant socio-economic causalities more accurately, Ekvall (1999) advocates using realistic default values for elasticities of supply and demand. A more extensive discussion of elasticities in relation to the multifunctionality problem is to be found in Section 3.9.

1.2.3.3 Spatial differentiation

Introduction

The reliability and validity of LCA results can be substantially improved by introducing spatial differentiation. Although location-specific data will rarely be available for all processes within a product life cycle, spatially differentiated assessment may be preferable for those processes for which the required information is available. Especially for processes that appear to predominate in the overall impact of a product life cycle, additional effort to gather location-specific information is advisable.

In recent years the issue of spatial differentiation has been gaining growing attention (*cf.* Potting & Blok, 1994; Pujol & Boidot Forget, 1994; Potting & Hauschild, 1997*b*; Tolle, 1997; Krewitt *et al.*, 1998, 1999;

Matsuno *et al.*, 1998, 1999; Moriguchi & Kondo, 1998; Nigge, 1998; Potting *et al.*, 1998, 1999; Schulze, 1999; Tørsløv *et al.*, 1999; Potting, 2000). No general, comprehensive approach has yet been developed, however.

To render spatial differentiation generally applicable to any process in any product life cycle, spatially specific equivalency factors are needed. Potting (2000) has made a start on developing such factors for the impact categories acidification and human toxicity. Her approach is geographically restricted to Europe, however, and the human toxicity model has been elaborated for two illustrative substances only. Moreover, the approaches adopted for the two impact categories differ with respect to how the ultimate effects are defined. The values of these exercises should not be underestimated, though. In the first place, they demonstrate that spatial differentiation is a worthwhile exercise, as there may be major variation among regions. In the second place, they form a good example of how spatial differentiation can be practically implemented in LCA.

A necessary first step towards the ultimate goal of constructing worldwide, mutually consistent equivalency factors for all impact categories is a systematic analysis and description of how such factors might be constructed.

Spatial differentiation in relation to the structure of Impact assessment

Heijungs & Wegener Sleeswijk (1999) distinguish three dimensions in life cycle characterisation of toxic (and other) substances: fate, exposure (or intake) and effect. The numerical impact of an emission can be considered to be composed of three independent variables, representing these dimensions. Spatial differentiation may be independently performed for each of these variables.

One of the most important starting points for Impact assessment modeling in LCA is a definition of the distinct character of emissions in the usual steady-state modeling type of LCA elaborated here, and a short analysis of its consequences. In normal LCA practice emissions are not considered as continuous fluxes but as discrete pulses, since they are linked to single amounts of product function, rather than to (production) processes as such. These discrete emission pulses cause discrete 'concentration pulses'. While continuous concentrations are characterised by concentration value ($\mu g/m^3$) and spatial magnitude (m^3 or m^2), concentration pulses are additionally characterised by their temporal value. To handle large numbers of such concentration pulses, it is convenient to integrate them over both space and time. Integration over space (in m^3) yields amounts. The double integration therefore yields *time-integrated amounts*. The advantage of space and time integration of emission pulses is that each pulse is thus characterised by a single value. This permits comparison and assessment of pulses having different spatial and temporal characteristics. The pros and cons of such integration, including the scope for making due allowance for the exceeding of thresholds, are discussed by Wegener Sleeswijk (in prep.).

Spatial differentiation of fate-related aspects

Multimedia fate models in Risk Assessment relate continuous emission fluxes to environmental concentrations (Mackay, 1991; Cowan *et al.*, 1995). These same fate models can be used in LCA to relate discrete emission pulses to time-integrated environmental amounts. For a given substance, emission compartment and emission magnitude, the magnitudes of these time-integrated amounts depend on degradation, immobilisation and multimedia transport: the three main aspects of fate.

Since multimedia fate models depend on geographical and climatological parameters, it is almost impossible to use them without applying some form of spatial differentiation. Parameters such as temperature, rainfall, soil composition and land/water surface ratio of continents may have a major influence on the multimedia distribution and/or degradation time of substances. The use of single values (point estimates, such as averages or default values) for such parameters can lead to major deviations between the modeling results and the real world. By distinguishing between geographical areas, the validity of the model can be improved, although this will lead to greater model complexity and additional data requirements.

Most existing multimedia models distinguish between at least three emission compartments: air, soil and water, and between at least four transport compartments: air, soil, water and sediment. Spatial differentiation with respect to substance fate amounts to further division of each of these *compartments* . Together, the compartments make up the so-called *unit world*. While the spatially non-differentiated unit

world consists of a (small) number of homogeneously mixed (sub-)compartments, the spatially differentiated unit world consists of a (larger) number of homogeneously mixed (sub-)compartments.

A fate factor is a parameter that connects a standard emission of a certain substance into a certain compartment to the resulting (time-integrated) amount of this substance in one of the compartments of the unit world. Since one emission to one particular compartment may be distributed over many other compartments as well, most emissions are linked to a large number of fate factors.

Spatial differentiation of exposure-related aspects

An exposure (or intake) factor is a parameter that relates a standard (time-integrated) amount of a substance in a single environmental medium or compartment to the relative amount of this substance that eventually becomes bioavailable for ('target') organisms via this medium or compartment. 'Exposure' in LCA terminology differs from the Risk Assessment concept 'dose' in two ways. In the first place, exposure refers to a discrete – rather than continuous – event. This is caused by the discrete ('mass loading') character of emissions in LCA. In the second place, exposure is a collective – rather than individual – measure: if the population exposed to a certain dose is doubled, the exposure itself is considered to be doubled as well. Here, LCA differs in terminology from Risk Assessment, where 'exposure' and 'concentration' are often used almost synonymously.

In toxicity analysis, separate *exposure modeling* is always necessary for species that are exposed to substances via different environmental media. For species that are (mainly) exposed via a single compartment, effect can be linked directly to the (time-integrated) amount of substance in this particular compartment by means of fate modeling. This is the case for water and soil ecosystems in models without spatial differentiation. In these models, the water medium and the soil medium each correspond to one compartment only. In the case of spatial differentiation, the sizes of the compartments into which each medium has been divided should be taken into account. For assessment with reference to single species – and especially for man – it will often be desirable to include population densities as well. For assessment vis-à-vis medium-wide ecosystems, it is difficult to say whether the volume of the compartment as such or a combination of volume and some sort of 'ecosystem density' measure is the best criterion for exposure assessment. For the time being, however, volume as such seems to be the most obvious criterion.

For man, the total exposure to a substance is often the resultant of exposure via different routes: inhalation of air, intake through the skin and ingestion of drinking water and food. Different exposure factors are required to link the (time-integrated) amounts of the substance in different media and compartments to the eventual exposure. Air is inhaled directly and (usually) unpurified. The air route is therefore relatively simple to model. Substances in water may reach us either via drinking water or via the consumption of fish and other sea foods. Exposure rates depend not only on the level of drinking water consumption, but also on fish consumption, on drinking water purification and on bioconcentration and biomagnification in fish. Substances in soil will seldom give rise to significant direct human exposure, with a few exceptions like inhaled dust particles from asbestos-reinforced roads. Indirect exposure via all manner of foodstuffs is a very significant exposure route for many substances, however. Exposure by this route depends on foodstuff consumption as such, but also on bioconcentration and biomagnification in all these foodstuffs, which are derived directly or indirectly from plants, including meat and dairy. Because of the enormous variety of foodstuffs we consume, the soil route is by far the most complex route for exposure modeling.

The most obvious aspect of spatial differentiation that needs to be elaborated in the context of exposure modeling concerns differences in consumption patterns. This applies not only to food but also to drinking water, which may be either groundwater or surface water, and either unpurified or purified in some treatment system. Consumption patterns should, if possible, be combined with the magnitudes of the populations to which these consumption patterns apply, in order to calculate overall worldwide exposure to a substance from each medium or compartment.

So far, spatial differentiation in exposure modeling has been fairly straightforward. It is further complicated, however, by the consequences of spatial differentiation in fate modeling. If media are divided into different compartments, it is not only the consumption pattern as such, but also the origin of foodstuffs, drinking water and even air consumed that determine eventual exposure. Especially for

foodstuffs, this is a complicating factor in the extreme, since large numbers of people no longer obtain their food from their immediate surroundings but from all over the world.

Spatial differentiation of effect-related aspects
An effect factor is a parameter that relates a standard exposure level for a species or ecosystem to a certain effect level. Sensitivity as such, the presence and abundance of sensitive species and background concentrations are effect-related aspects.

The concept of sensitivity is intimately related to the dose-response relationship. A numerical representation of sensitivity may be based either on the exposure value at which a species starts to show adverse effects to a substance (NOEC or NOAEL in toxicity assessment) or on the relative magnitude of the response to a standard increase of dose in the response area (the slope of the dose-response curve at the prevailing dose level). For doses in the area of the curve where the slope is greater than zero, the second possibility seems to be by far the best representation of sensitivity. For doses below no-effect levels, however, it is hard to say whether the relative distance to the point where adverse effects start to occur, or the (average) strength of the response to an increase of dose in the response area is the greater determining factor with respect to the latent, or 'potential' effect. For some substances, no-effect toxicity levels are lacking, for carcinogenic organics and for radioactive particles, for example. The dose-response curve of such substances has an F-shape, instead of the familiar S-shape. Similar dose-response relationships may hold for non-toxicity-related impact categories. For toxicity, it is common practice in LCA to work with no-effect levels, rather than slopes, as measures of toxicity. For substances lacking real no-effect levels, threshold values may instead be used. For other impact categories, it is more common to work with values more or less representing slopes, although real dose-response curves are seldom used in these contexts. Whenever dose-response curves are used, it should be borne in mind that the relationship between the qualitative concept of 'response' and the numerical values representing this concept should be explicitly defined.

Both the no-effect level and the slope of the dose-effect curve vary per species and per substance. For aquatic and terrestrial ecosystems, no-effect levels have been defined by combining no-effect levels for various subgroups of species in the respective ecosystems. Although different tests often show major variations in sensitivity among representatives of a single species, these variations can seldom be attributed unambiguously to location-specific conditions. Contrary to the sensitivity of single species, the sensitivity of ecosystems may show important spatial dependencies, however. This is not due to sensitivity as such, but to the presence and abundance of sensitive species in given regional ecosystems: in areas lacking sensitive species, the effects of an emission will be less readily observable than in areas where such species are abundant. This phenomenon constitutes an important assumption in the critical load concepts for acidification and eutrophication. In the case of human toxicity it is included by taking population density into account. In the toxicity model used for calculating ecosystem no-effect levels, however, the regional presence and abundance of sensitive species has not yet been taken into account.

Since dose-response relationships are seldom linear and homogeneous, the dose-response ratio is not independent of background concentrations. In regions with low background concentrations, effects may not occur, despite the presence of sensitive species. It is a matter of choice whether the purely potential effects in such 'below-threshold' areas are taken into account. Especially for naturally occurring substances such as minerals, which may even be benevolent in low concentrations, this choice will not always be easy to make. In 'above-threshold' areas, background concentrations determine the prevailing slope of the dose-response curve. In more refined models, this may be taken into account. The relevant degree of spatial differentiation must then also be incorporated in inventory modeling and fate modeling.

1.2.3.4 Average and marginal processes and average and marginal process data

Basic concepts
The basic mathematical meaning of the terms 'average' and 'marginal' is clear and is summarised below. Still, application of these inherently clear concepts may give rise to difficulties. Implied in an LCA-relevant choice is the (exogenous) change in the volume of some product, as produced by some unit process (plant, facility, etc.). How, then, can we describe the resultant changes in the other inputs and

outputs of that process? Let us simplify the situation to a process producing one functional output (x, say, electricity) and that has one other flow (y, say, input of fuel). A change in the demand for electricity will induce a change in the production volume of the generation process.[1] This can be considered as a shift from the reference value of x (which we denote by x_0) to the value of x implied in the choice for alternative 1 (for which we use x_1). We shall call this an incremental change. The associated incremental change in fuel demand (y) is from y_0 to y_1.

The relationship between x and y is known in economics and engineering as the production function.[2] It is a result of the interplay between physics, chemistry, technology and economics.[3] The relationship can be symbolised as a function f which maps any value of x onto a value of y (or the other way around). The production function is thus

$$y = f(x) \qquad (1.2.3.4.1)$$

In general it is a non-linear function, involving many variables. We here treat the case of two variables with known relations.

In the case of an incremental change of electricity demand Δx (from x_0 to x_1) we need to calculate Δy (from y_0 to y_1). This means we have to calculate

$$\Delta y = f(x_1) - f(x_0) \qquad (1.2.3.4.2)$$

Our general ignorance regarding the production function f already makes this calculation problematic. A second problem is that we cannot simply rewrite this equation as a function of the change in x, as $\Delta y = g(\Delta x)$, say, even if $f(x)$ is known[4].

There are, however, two situations which permit important simplifications to be introduced. One is referred to as a marginal change, the other as a change for which average[5] data may be appropriate. See Figure 1.2.3.4.1.

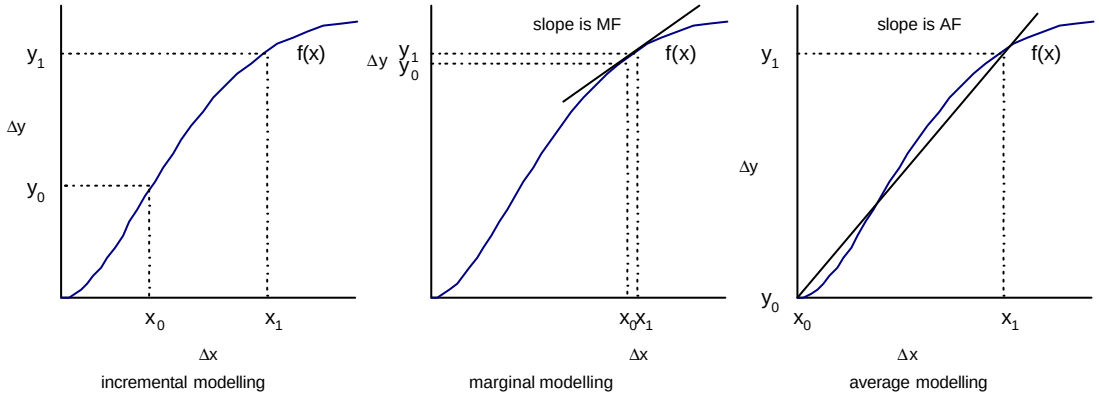


Figure 1.2.3.4.1: Incremental, marginal and average analysis of a relation between two variables.

If the change is fairly small (as when drawing 40 W more from an existing, large power plant), we may make a linear approximation of the non-linear production function. In that case, we use

$$\Delta y = MF \times \Delta x \qquad (1.2.3.4.3)$$

where MF is the marginal factor (the 'slope') for the input of fuel.[6] This formulation is known as using marginal data, because the data apply to a marginal change. It has the major advantage that one value,

[1] For simplicity, we concentrate here on the direct (short-term) change. In the longer term such a change will lead to changes in maintenance, replacement, investments and so on. This issue will be expanded on in a later section.

[2] In line with the previous note, there exist short-term and long-term production functions.

[3] Supply relations for inputs and demand relations for outputs are omitted from the analysis here. In combination with a given production function, these will determine the output change due to an exogenous change.

[4] Only if the function f is linear through the origin or, more generally, linear and homogeneous to degree 1, can this equation be used.

[5] The terminology is confusing because it differs depending on context. In economics, the relevant context here, long-term marginal cost equals the short-term average for a given technology.

[6] More precisely: it is the first derivative of f with respect to x, evaluated at $x = x_0$. There is one formal requirement for this approximation: f must be differentiable, which means that it must be continuous. In

MF, suffices to calculate the change in fuel input for any change in electricity production, as long as that change is small enough to justify linearisation.

If, on the other hand, the change is 'revolutionary' in the sense of resulting in a complete shut-down of a facility ($x_1 = 0$) or a full new start-up ($x_0 = 0$), a different approximation may be used:

$$\Delta y = AF \times \Delta x \qquad (1.2.3.4.4)$$

where *AF* is the average factor for the input of fuel.[1] This approach is known as using average data. Here, there is no such thing as a corresponding average change. Its name derives from the fact that at the non-closed working point, the average fuel required per unit electricity generated coincides with the average factor. Using the average factor again has the major advantage that one value, *AF*, suffices to calculate the change in the fuel input for any change in power production, as long as that change is 'revolutionary' enough to justify complete start-up or shut-down.

Different meanings of 'average'

In the above, we introduced the term 'average' as a means of dealing with process data in 'revolutionary' changes. We did not mention it with respect to incremental or marginal changes. However, it may be argued that using marginal data for ascribing interventions to activities may introduce an element of unfairness. If a train is carrying 100 passengers, one can calculate the average power consumption per passenger. If one extra passenger enters the train, the train's power consumption will increase marginally. There are, now, two alternative principles that can be adopted for assigning power consumption to this extra passenger:

A. the 'factual' approach: the marginal change in power consumption is assigned to the extra passenger, or
B. the 'fair' approach: the additional power consumption is distributed evenly among all 101 passengers.

The second approach has the advantage that historical facts (such as who was the last passenger) are irrelevant: every passenger is treated the same way and is 'responsible' for a proportional[2] amount of electricity use and associated environmental interventions. This also means that this approach is not susceptible to 'strategic abuse', as when someone argues that the aircraft was flying anyhow, forgetting that structural changes may be induced by occasional choices taken by many individual people. [3] It also has its disadvantages, however. One problem of a practical nature is that one can envisage different methods for establishing partitioning over the one hundred passengers: based on number (every passenger 1/100th), on mass (with or without luggage), on share in sales (first class versus second class passengers), etc. There is a more fundamental problem, though. To some extent, the use of averages seems to be at variance with the original idea that LCA is supposed to provide a model of what is actually happening. The short-term marginal indicates what is happening first, while the long-term marginal indicates what will ultimately happen in a steady state (where the marginal is equal to the average for a given technology). Integrating a dynamic model would cover the long and the short term, but is beyond practical operational feasibility. We choose the long-term average as the most fitting approach for structural decisions, neglecting short-term mechanisms in the detailed LCA. Such elements may be introduced in the extensions to detailed LCA, however.

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practice, *MF* may be recorded in a database, even without full knowledge of the function *f*. *MF* is then obtained through a small but non-zero change around the typical working point of the process.

[1] More precisely, in the case of start-up it is the ratio y_1/x_1 and in the case of shut-down the ratio y_0/x_0. When the non-closed working point of the production characteristic in Figure 4 is the typical working point of the unit process, the two cases start-up and shut-down will yield the same average factor. There is one formal requirement here: the closed working point should have coordinates (0, 0), which means real closure. A boiler that is switched on but not used still requires some energy, but a boiler that is switched off does not. We do not preclude any choice in the treatment of capital goods here.

[2] This is one reason for preferring the term 'proportional' to 'average'.

[3] One can also avoid ethical-sounding terms like fair by stating the problem slightly differently, as a situation A with 100 passengers, with 1/100th part of this total power consumption allocated to each passenger on average, and a situation B with 101 passengers, with 1/101th part of total power consumption as the average allocation. However, it should be remembered that, for structural decisions like most of those examined here, installed capacity is adjusted to a change in output while intended capacity use (or operating point) remains the same. The extra passenger then requires 1/100th of an extra train and the average remains the same and only situation A is relevant.

Whichever of the two approaches is preferred, an important lesson here is that even marginal data may be used for average (= proportional) assignment. Fortunately, the dichotomy between factual and fair partitioning is not important for the structural type of modeling developed here; see our recommendations at the end of this chapter.

Different meanings of 'marginal'

Next, there is the problem what exactly is on the axes. First, let it be clear that a production function is normally more than a relation between one input and one output. It may be a whole vector of inputs that is related to a whole vector of outputs. Some pairs of inputs or outputs may be directly coupled (as in the case of burned fuel and generated electricity), while other pairs of inputs and outputs may be related only in a longer time perspective (as in the case of replaced generators and generated electricity). A marginal change, such as the effect resulting from changing one unit, may thus be modeled with different time frames in mind. In economics, for example, the short-term marginal effect of increasing production equals the change in variable costs, as in the case of the extra petrol needed to get an extra passenger airborne each flight. In the somewhat longer run, an extra passenger will lead to extra food on board, extra cabin maintenance, etc., so the marginal costs will come to exceed the variable costs. In the still longer run, the number of flights will rise, as scheduled flights are related to percentage occupancy. Ultimately, extra aircraft will be built. Therefore, the marginal costs will now include those of the extra aircraft built and operated to adapt to the increased demand. For a given type of aircraft, the long-term marginal costs hence equal the short-term average costs. In the further chains, similar sequences result, leading to the opening of extra bauxite mines, extra cargo ships, etc. These long-term marginal costs, as required in LCA for structural decisions, are effectively equal to the average costs, as these reflect the costs of all fixed overheads. The lesson here is to be precise in the uniqueness of the decisions studied. In the aircraft costs example, there was a subtle change from one passenger extra once, to short-term effects of one passenger extra on each flight, to longer-term effects of one passenger extra on each flight. Each shift means that a different operational meaning of the adjective 'marginal' is to be understood.

Finally, we must distinguish between the marginal process and the marginal data of a process. There may be one (or more) unconstrained process(es) that can be identified as the process(es) that will be used to satisfy the extra demand for a certain product. This process is the marginal process in a process mix. Next, it can be determined how emissions, fuel requirements, etc. will change when one marginal unit of output is needed. These changes then reflect the marginal process data, in the short, medium or long term. It is perfectly possible (or it may even be necessary) to treat the marginal process using long-term average data, so the two concepts should be clearly distinguished.

In change-oriented LCA, the analysis is concerned with incremental changes rather than averages. However, incremental changes, like marginal changes, may coincide with average effects under the assumptions stated above. These assumptions apply in long-term modeling, as required for the analysis of structural changes.

In situations where a really simple model is used, these analytic differences may disappear. With linear relations not passing through the origin, marginal and incremental effects coincide, but differ from average effects. With linear relations through the origin, marginal, incremental and average effects are all equal. Thus, in very simple models the discussion on average and marginal data may be avoided, but not the discussion on the marginal process.

1.2.3.5 Quantitative and qualitative information and notation

In principle, LCA is interpreted in this report as quantitative life cycle assessment. The model discussed so far concerns quantifiable flows of products, materials and energy, quantifiable emissions and resource extractions, and quantifiable contributions to environmental problems. It may seem unnecessary to discuss exactly we mean by 'quantitative information'. However, it may be rewarding to be somewhat more explicit and also to include a few words on the scope for including qualitative information.

First, an example of a purely qualitative LCA. Suppose that we adhere to a dogma that says: no tropical hardwood. This would mean that tropical hardwood window frames are bad. In a life cycle context, however, it would mean that aluminium window frames are equally bad, since there will be tropical hardwood somewhere in this life cycle as well. Perhaps the office of the accountant of the aluminium smelter has tropical hardwood window frames. Of course, the amount of tropical hardwood per functional

unit of window frame is much lower for aluminium window frames than it is for those made of tropical hardwood. If we wish to take this into account – and obviously we do – we must enter the domain of quantitative analysis. We conclude that qualitative information like 'contains hardwood' is difficult to employ in a comparative analysis covering more than a single aspect.

A more sophisticated case is the desire to include qualitative attributes of the quantified information. An example is given by comparison of tropical hardwood window frames with and without an FSC certificate. One could argue that this calls for a method that is capable of handling qualitative information. Better still though, we could also recognise a parallel with distinguishing between emissions of copper from mercury, and distinguish extractions of FSC-certified wood from non-certified wood. In other words, the primacy of quantitative information does not imply that we stop using language to indicate the identity of the flows modeled. Technically, every distinction within a class of flows means that these flows are regarded as different. Thus, if we distinguish certified from non-certified tropical hardwood, we actually have two different flows for hardwood. Likewise, distinguishing atmospheric and aquatic releases of lead effectively yields two different flows that have the term 'lead' in common. And the same applies to every form of spatial and temporal differentiation. It is a matter of personal preference whether we mentally consider Iberian SO_2 and Scandinavian SO_2 as two different flows or as one flow that comes in two varieties. In terms of modeling, however, they are two different flows, with separate entries in data tables.

The general structure for indicating quantifiable entities in any science is a tripartite one (see, for example, the Handbook of Chemistry and Physics):

'quantity' = 'value' × 'unit'

(not to be read as an equation), as in

the temperature is 23 degrees Celsius.

We thus have the following elements:

- quantity: the quantifiable entity of interest, like temperature;
- value: the numerical magnitude, like 23;
- unit[1]: the yardstick used to express the value, like degrees Celsius.

This same general structure is, in principle, applicable to the LCA model, including its input and output data. However, several remarks are in order.

- 'Quantity' is a confusing word, because it also indicates 'amount' ('quantity of product required for the functional unit'). Other terms are 'variable', 'parameter' and 'flow type', and, in specific contexts, 'observable', 'data', and 'coefficient'. All these terms also have disadvantages. In the context of LCA modeling the term 'variable' seems to be the most appropriate. Observe that the quantity may be a simple one, like temperature or emission of SO_2, but it may also be more complex, like sea temperature or emission of SO_2 in Sweden.
- 'Value' is probably an even more confusing term than 'quantity', because it may easily suggest a focus on economics, welfare or utility. The term 'numerical value' is a rather clumsy solution. 'Magnitude' may also indicate the type of variable rather than its size. 'Size' and 'amount' are probably more unequivocal and neutral terms.

[1] The terms 'unit' and 'dimension' are sometimes used incorrectly as synonyms. Every quantity has a unique dimension. Speed, for instance, has the dimension length over time. Dimensions therefore provide no yardstick. Units are specific cases of yardsticks. The dimension 'length' can be expressed by the units meter, inch, mile and so on. Moreover, different quantities may share the same dimension. For instance, length, distance, height and thickness all have the dimension length. Finally, certain quantities can have a composite dimension with a non-composite unit. Energy has the dimension mass times length squared over time squared, and is usually expressed in joule (J), although the composite unit kg times square metre over square second is identical with joule.

– Even the term ‘unit’ is not free of contextual differentiation, as in, for instance, ‘unit processes’ and ‘functional unit’. Furthermore, ‘units’ in LCA are often used loosely. The *Système International des Unités* lists basic and derived units, like kg, m and m/s. In published LCAs, guidebooks and software, one often sees unconventional units, like ‘kg/functional unit’, ‘GWP’ and ‘mPts’, and units may even be lacking altogether.[1]

As one terminology for all the disciplines involved in LCA seems too much to ask for, we here simply state our first preference, with what appear to be relevant synonyms:

Variable:	parameter, quantity, entity, flow type, flow
Value:	amount, size (possibly magnitude)
Unit :	[no synonyms]

Finally, variables are often abbreviated to user-defined symbols, while units are abbreviated using conventional symbols. Speed, a variable, is written as *v*, *u*, *s,* or some other symbol. Attributes may be indicated using other symbols, subscripts or diacritical symbols, like *v* versus *u*, v_{inside} versus $v_{outside}$, or *v* versus *v'*. It is conventional practice to italicise symbols. A possible unit for speed is metre per second, abbreviated to non-italic m/s. Values are not abbreviated, although the unit may be subdivided or expanded using standard prefixes, as when 1200 m is written as 1.2 km. Given the prominent usage of mathematical models for analysing environmental impacts, it seems only natural that the conventions of mathematical notation developed in physics should be adopted in LCA, too.

In addition to qualitative attributes that can be regarded as part of or an attribute to the flow name, it may be the case that we have real qualitative data. An example is a unit process which is known to disperse an ‘awful odour’. If such information is not quantifiable and we wish to keep this information available throughout the LCA, we must add it to the other qualitative and quantitative aspects of the unit process, and carry it over to the inventory table and subsequent Impact assessment and Interpretation. In the example with which we opened this section (the phrase ‘contains tropical hardwood’) we showed, however, that this type of purely qualitative information is of very limited value in the course of an LCA.

[1] What units would we recommend in these cases? “Kg/functional unit” is sometimes seen in inventory tables for the aggregated emissions of one functional unit. Almost all calculations are about a functional unit, so adding this is superfluous, just as it is superfluous to say that a can of paint contains 0.75 litre *per can*. A mere “kg” suffices here. In the “GWP” example more is involved. The GWP (global warming potential; see section 4.3.5) is a quantity (or variable) and it is an inherent (or intensive) property of a substance that does not depend on the amount of substance. One can speak of the GWP of methane, just as one can speak of the density of stone. The GWP of methane is a dimensionless 21, and the multiplication of inventory results (in kg) by their respective GWP yields a quantity that is again expressed in kg. “mPts” or “millipoints” is a unit that is sometimes used for the weighting result after characterisation, normalisation and weighting. If characterisation yields category results in kg and normalisation totals are in kg/yr, normalisation results are in yr. Finally, if weighting factors are dimensionless, the weighting result is again in yr. If the reference flow of the functional unit is expressed in terms of “function years”, normalisation and weighting results are dimensionless.

1.2.3.6 Reporting of LCA

Reporting an LCA is an important issue. Because the methods for LCA are manifold, databases often give conflicting figures, and many choices must be made in the course of completing an LCA, the results are highly dependent upon the exact details regarding methods, data and choices. The reporting of an LCA therefore implies far more than reporting the results of the LCA. It also embraces reporting the methods used, the data used, the choices made, the procedural context in which the LCA was produced and the exact question and purpose of the LCA.

In the guidelines that accompany the operational steps implemented in this Guide, reporting guidelines will be given separately for each step. Equally important, however, is that the person or persons reporting an LCA develop an awareness of certain basic principles of LCA reporting. ISO 14040 (1997E) clause 6 reads as follows concerning the topic of reporting requirements (see textbox):

> The results of the LCA shall be fairly and accurately reported to the intended audience. The type and format of the report shall be defined in the scope phase of the study.
> The result, data, methods, assumptions and limitation shall be transparent and presented in sufficient detail to allow the reader to comprehend the complexities and trade-offs inherent in the LCA study. The report shall also allow the results and Interpretation to be used in a manner consistent with the goals of the study.
> When the results of the LCA are to be communicated to any third party, i.e. interested party other than commissioner or practitioner of the study, regardless of the form of communication, a third-party report shall be prepared. This report constitutes a reference document, and shall be made available to any third party to whom the communication is made.
> The third-party report shall cover the following aspects:
>
> a) general aspects:
> 1) LCA commissioner, practitioner of LCA (internal or external);
> 2) sate of report;
> 3) statement that the study has been conducted according to the requirements of this International Standard.
> b) definition of goal and scope;
> c) life cycle Inventory analysis: data collection and calculation procedures;
> d) life cycle Impact assessment: the methodology and results of the Impact assessment that was performed;
> e) life cycle Interpretation:
> 1) the results;
> 2) assumptions and limitations associated with the Interpretation of results, both methodology and data related;
> 3) data quality assessment.
> f) critical review:
> 1) name and affiliation of reviewers;
> 2) critical review reports;
> 3) responses to recommendations.
>
> For comparative assertions, the following issues shall also be addressed by the report:
> – analysis of material flows to justify their inclusion or exclusion;
> – assessment of the precision, completeness and representativeness of data used;
> – description of the equivalence of the systems being compared in accordance with 5.1.2.4;
> – description of the critical review process.
>
> Source: ISO 14040, 1997.

Lindfors *et al.* (1995a) and the SETAC-Europe Working Group on Case studies (Meier *et al.*, 1997) give additional material for basic and specific guidelines for reporting. As general principles, we propose the following guidelines:

– The results of the LCA shall be fairly and accurately reported to the intended audience. The type and format of the report shall be defined in the scope phase of the study.

- The result, data, methods, assumptions and limitation shall be transparent and presented in sufficient detail to allow the reader to comprehend the complexities and trade-offs inherent in the LCA study. The report shall also allow the results and Interpretation to be used in a manner consistent with the goals of the study.
- When the results of the LCA are to be communicated to any third party, i.e. interested party other than the commissioner or practitioner of the study or to the expert reviewer, regardless of the form of communication, a third-party report shall be prepared. This report constitutes a reference document, and shall be made available to any third party.
- A third-party Goal and scope report and a third-party Final report are mandatory for detailed LCA. For simplified LCA, only a Final report is mandatory; it is recommended that this Final report also conforms to the guidelines for third-party reports given below.
- The terminology used in the report shall be consistent with the ISO 14040 series. Other terms used shall be detailed and defined and used consistently throughout the study.
- If, in principle, the LCA study undertaken conforms to the guidelines given in this document, any deviations from the guidelines provided shall be specifically documented and justified fairly and transparently.
- For each LCA phase and step, all assumptions and value judgements shall be clearly detailed, along with the justification for these assumptions.

Since in this report some steps are named and sequenced a little differently from the ISO standards (see Chapter 1), the ISO reporting issues have not been adopted unaltered. All the ISO issues are covered, however, in the reporting guidelines given above. Furthermore, where possible and useful the ISO guidelines have been made more explicit, based on the guidelines provided by Lindfors *et al.* (1995a) and Meier *et al.* (1997).

A distinction is made here between reports and reporting issues. Two main reports are distinguished: the Goal and scope report and the Final report. There is a relation between reports and issues to be reported, of course. For example, if it is decided to also draft an Inventory and/or Impact assessment report, all issues should be reported concerning that phase and the preceding phases.

For a Goal and scope report, however, a different list of reporting issues holds. The reason for this is that in the Goal and scope phase of a study, the study results are not yet available and so cannot be discussed, evaluated, etc. In contrast, the initial choices and assumptions proposed can be reported and discussed at an early stage of the study with the commissioner and third parties, i.e. interested parties and/or expert reviewer. A Goal and scope report is particularly recommended for detailed LCA studies, since one main characteristic of detailed LCA studies is that some kind of critical review is made for which a third-party report is necessary and in which case it is useful to discuss the main choices and assumptions at an early stage of the study. By allowing comments at an early stage, the LCA study may get off to a more efficient and smoother start.

The resulting Goal and scope report may be used as the basis for the Final report. Of course, modifications made to the former report during the course of the study should be duly documented and justified. When a full description of all methodological issues is provided in the Goal and scope report, consideration may be given to reporting only the results in the chapter of the Final report on Inventory analysis, Impact assessment and Interpretation.

In the Goal and scope report the practitioner can justify the choices and assumptions made for each step by basically referring to the Guidelines given in this report or explaining why and how (s)he has deviated from these Guidelines. Thus, the goal and scope report can be a short report referring to the Guidelines as defined in this Guide, or a more extensive report explaining why and how other choices and assumptions (including extensions) have been made. On the one hand, extensions can be made to the detailed Guidelines. On the other hand, even the Guidelines for simplified LCA can be further simplified. However, it should then be clearly documented if and how these further simplifications are to be justified in relation to the goal of the study.
For simplified LCA we recommend not writing a specific report as part of the Goal and Scope definition phase. In a simplified LCA only a Final report should be written.

1.2.3.7 Software for LCA

Scientific endeavours to develop a method for LCA lead to calculation procedures involving a vast multitude of input data. Today it is not unusual to find 500 unit processes, 600 economic flows and 1000 environmental flows in a typical LCA. Clearly, such an analysis cannot be undertaken by hand and calculations must therefore be run on computers. There are three ways to use computers:

- One can use a general, i.e. non-LCA-dedicated program such as a spreadsheet. This has certain advantages: most (potential) practitioners already know how to work with these programs, they are quite flexible in manipulating data and they provide easy access to graphical presentations. On the other hand, the open and flexible nature of these programs make them rather inefficient for this particular purpose: the user must in fact himself create all the links between flows and processes and the same applies to the computational algorithms.
- One can also use a dedicated LCA program. There are several dozen such programs available, ranging in price from free to many thousands of euros and in scope from very basic to quite advanced. Many commercial programs are shipped with a database of process data and/or Impact assessment data. A clear disadvantage of commercial software is that the user has no capacity to perform analyses deviating, however slightly, from those that are part of the program. Some programs cannot deal with allocation, while others can handle one allocation method only. There is currently no program available that provides for full implementation of all the methods specified in this Guide.
- As a compromise between the flexibility of spreadsheets and the power of dedicated programs, a practitioner may decide to develop his own LCA software. This is a major task. This is clearly not an option to be recommended as standard practice.

For the occasional practitioner dealing with very small product systems (say, less than 20 unit processes) we tend to recommend the use of spreadsheets. As product systems grow or if detailed analyses are required, including sensitivity and uncertainty analyses, for instance, the use of commercial software is advisable. We refer to Rice (1996), Rice *et al.* (1997), Menke *et al.* (1996) and Siegenthaler *et al.* (1997) for an overview of the programs then available[1].

1.3 Management of LCA projects: procedures

1.3.1 Introduction

In principle, an LCA is an analytical activity that should be performed by independent experts. However, LCA projects also generally involve using the analytical results within a policy or strategy framework, as is the case with the majority of LCA studies conducted or commissioned by public authorities or private companies. In these cases the results of the LCA will have an influence on government or corporate decision-making and we then speak of mandated science: a scientific, analytical activity is performed for which a mandate has been given, and the outcome may steer decision-making.

Authoritativeness of results

ISO 14040 defines an interested party as “an individual or group concerned about or affected by the environmental performance of a product system or by the results of an LCA”. While retaining this definition, in this Guide we generally employ the shorter synonym ‘stakeholder’. Three groups of environmental stakeholders are usually recognised: political (national, international legislators), public (press/media, local environmental initiatives and consumer and environmental organisations) and market (competitors, customers, suppliers and financial institutions).

When LCA is applied as mandated science problems of authoritativeness often arise, that is to say: the outcome of the LCA may not always be accepted by all the stakeholders in the policy or strategy. The upshot will be clear: if the outcome if not accepted, the LCA will be of no influence on decision-making. Problems of authoritativeness may arise for one of three reasons:

- the actual results of the LCA may be debatable, owing to dubious assumptions, data and/or system boundaries being used, for example;

[1] A program that is intended mainly for educational purposes has been developed at CML. It can be downloaded, for educational purposes for free, at: http://www.leidenuniv.nl/interfac/cml/ssp/cmlca.

- there may be a misfit between the LCA results and other considerations pertinent to decision-making (safety, cost-effectiveness, etc.);
- the policy/strategy setting often involves many different parties representing differing interests. In such cases, parties will endeavour to magnify the above objections, for reasons of strategy as well as substance.

Conducting LCAs in accordance with ISO standards

If an LCA project is performed according to ISO standards, this means not only that the LCA itself will be methodically structured but also that certain aspects of the LCA process will be established beforehand. There are two important issues here. First, the ISO standards lay down (quality) criteria for the design and execution of the LCA as such as well as for the reporting of results, data, methods, assumptions and limitations. Second, the ISO standards outline a procedure for a 'critical review'. In general terms, the ISO standards deem a critical review optional and indicate that use can be made of different review options. If the LCA results are used to support 'comparative assertions' a critical review is mandatory ("since this application is likely to affect interested parties that are external to the LCA study") according to § 7.2 of ISO 14040. In cases involving "a comparative assertion that is disclosed to the public" a "review by interested parties" is required under ISO 14040 (clause 5.1).

For the sake of clarity, we here define several other key terms used in this Guide. An 'LCA study' is an environmental study in which LCA methodology is employed, performed by practitioners who may or may not be affiliated to the party or parties commissioning the study. An 'LCA project' is a project that seeks to obtain particular results by means of an LCA study. Besides commissioning parties and practitioners, the project may also involve other organizations and individuals, in the capacity of data supplier, peer reviewer or interest group, for example. An 'LCA process' is the integral series of exchanges among the individuals and organisations participating in an LCA project, from project initiation and guidance through to interpretation and discussion of the results.

Use of a process approach

Against this background, LCA-based decision-making can be seen as a process designed to involve all relevant stakeholders, which may take a variety of forms. This implies a need to elaborate a process approach, with the process being designed as appropriately as possible for the specific nature of this kind of decision-making and the various specific situations that may be involved. If a process approach is successfully implemented, the process result (i.e. the outcome of such an approach) will show a number of characteristics.

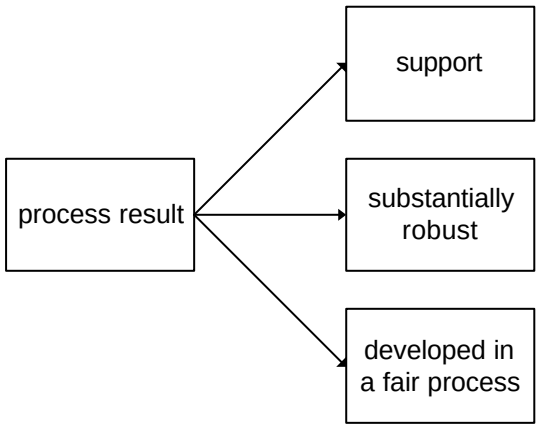


Figure 1.3.1.1: Result of the process approach (De Bruijn *et al*., 1998)

In the first place, there will be due support for process result. Having exerted an influence on the results, the various stakeholders will often come to hold different ('richer') views. Second, the process result will be substantially robust. That is to say: the outcome will be scientifically so well underpinned as to stand up to criticism. The stakeholders will have contributed their know-how and information and enriched the results with their knowledge and values, with due allowance being made as far as possible for the dynamics of new developments, innovation and so on. Third, the process will have been fair. All the

stakeholders will have been able to contribute their problem definitions and solutions. These will all have been taken into due consideration in the decision-making process and a decision ultimately reached. This process will have been transparent, allowing it to be validated by all parties.
Together, (1) support for a (2) substantially robust result after (3) a fair process may be sufficient to ensure that there is consensus among the stakeholders. In many cases, though, the interests of the stakeholders may differ so widely that consensus cannot reasonably be expected. A process may then well result in commitment to a particular product. Parties expressing commitment declare that they:
- commit themselves to the product of a process;
- perhaps distance themselves from certain elements of that product;
- are prepared to vouch for the product.

1.3.2 Decision-making based on a process approach

The philosophy behind the process approach is that the outcome of a LCA will be authoritative only if the principal parties are duly involved in this analysis. This involvement should be structured in an orderly fashion and a process design is consequently essential. This design should indicate:
- which parties (singular or plural) are to be involved in the analysis;
- at which points in the decision-making process these parties may exert their influence;
- how to proceed at such points in the process.

Advantages
In brief, the principal arguments in favour of a process approach are as follows:

I. Support
Successful execution of an LCA requires the support of the stakeholders, and if they are to stand behind the outcome of the analysis they must be duly involved in the analysis process.

II. Quality of data and other information
The stakeholders can also make an important contribution to the quality of the information used to perform the LCA, for they dispose over factual data and other information (based on such data), in the form of data and know-how on recent and expected innovations, for example.

III. Quality of the analysis
Critical interrogation by stakeholders vis-à-vis the study and its outcome will bring into clear view the underlying values, the choices (regarding data and assumptions) made, and which results are robust and authoritative and which do not satisfy these criteria.

IV. Validation of stakeholder views and assumptions
Conversely, in their meetings with researchers the stakeholders will have to expound on their own views and assumptions, some of which may not stand up to the scrutiny of scientific criticism.

V. 'Enrichment'
The stakeholders involved in the analysis will often be representing different interests. Confrontations between these interests may lead to enrichment: a mutual learning process.

Basic preconditions
If a process approach is to have any chance of success, three basic preconditions must be satisfied:

Condition 1: There must be a sense of urgency.
A process design demands a sense of urgency. The stakeholders must, in other words, be convinced that:
- there is a problematic situation[1] that must be resolved as a matter of priority;
- an LCA can help resolve this problematic situation;
- the stakeholders must somehow collaborate on designing the LCA.

If these three conditions are not fulfilled, the process will have little chance of succeeding, for nobody will be prepared to commit themselves to a decision-making process. Phenomena like the 'participation paradox' as well as limited participation at the outset of the process (see Section 1.3.4) are especially

[1] The stakeholders may well, at the outset, have different perceptions of the problem. What matters, though, is that all parties see the need to arrive at a collective approach and decision.

likely if there is little sense of urgency. There is then a high major risk of the process breaking down.

Condition 2: Stakeholders must be willing to commit themselves to a process design.
It is important that the basic process agreements between parties are explicitly recorded in a process design, making clear which organisations may participate in the process, who are to represent them, what mandate these representatives are to have, and what rules of decision-making and what (substantial, financial) constraints are to hold. Although this may seem trivial, experience shows that process management is frequently implicit: talks are held with stakeholders, without a process design being properly thought out and elaborated in concreto. The transparency and integrity of the process may consequently suffer. Parties may be drawn into the process too late, be unclear about the status of the talks and how their contribution is reflected in the end result, have the impression that other parties are exerting undue influence, and so on. If there is a lack of transparency and/or integrity, a process may become 'messy' in the eyes of the participants and/or not give them a fair chance to influence the outcome. It will be clear that this will discredit the process.
The process design can be drawn up by a process architect, in due consultation with all the stakeholders. Once the latter have approved the design, the process can be steered and guided by a so-called process manager.
A number of topics described in the following sections under 'process items' derive from this fundamental precondition.

Condition 3: All process participants must possess a minimum level of substantial expertise.
A third basic condition is that the representatives of the parties must be in possession of a minimum level of knowledge and expertise on LCAs. If this is not the case, pronounced asymmetry may arise among process participants, which frequently leads to the process breaking down. The mere fact that participants lacking due expertise will need to go through a learning process during the LCA process confirms this risk: the process will become unnecessarily protracted and consequently vulnerable.

1.3.3 Designing the decision-making process

The first thing to remember when designing a decision-making process is that a process approach in a situation of mandated science involves certain risks. A number of such risks can be identified and for these due allowance should be made.

- stakeholders will have access to knowledge regarding project progress and may make opportunistic use of interim and final results;
- if the LCA results threaten to be unfavourable from their perspective, stakeholders may so accelerate or procrastinate decision-making that a misfit ensues (substantive, analytical results arrive too soon or too late);
- researchers have access to information about the progress of the decision-making process and may upset this process by publicising their opinions, incomplete ('quick and dirty') research results, and so on;
- if the study results threaten to be unfavourable from their perspective, researchers, too, may so accelerate or procrastinate decision-making that a misfit ensues; at the very least, the stakeholders may be accused of taking a decision that is inadequately grounded in the research undertaken;
- stakeholders and researchers may be so concerned about reaching consensus that the quality of the LCA suffers as a result.

These risks are a clear indication that an explicit process design is a sine qua non for a successful process-type LCA, for it allows appropriate measures to be taken to limit the risks inherent in such a process.

Focus on execution and use of the LCA

In the second place it should be recognised that fair LCA-based decision-making requires that due allowance be made for how the LCA is conducted and how the analysis results are utilised by the various stakeholders. On the one hand, this relates to the actual analysis based on the LCA methodology. In this respect, the main aim of the process approach is to make sure the analysis is of sufficiently high quality and the results sufficiently authoritative for all the stakeholders. On the other hand, it relates to the practical implications of the analysis results, in the form of corporate or public decision-making, for example. In this respect, the process should be designed such that due justice is done to the analysis results in the corporate or public decision-making setting in question. This implies that the process design must cover more than merely the methodological rigour of the LCA project. For a process design to work satisfactorily there must also be optimum interaction between project execution as such and the intended practical use of the analysis results.

Matching the process goal

Thirdly, the process must be designed in accordance with the goal. The reasons for conducting an LCA differ from case to case. For example, a company may perform or commission an LCA in order to decide - internally - what kind of new product is preferable from an environmental perspective, or to provide public accountability for the environmental burden caused by a current product. Alternatively, it may be a government commissioning or conducting an LCA, for any number of reasons. The study may serve to decide - internally - what kind of new policies are preferable from an environmental perspective, or to provide public accountability for (proposed) policies.

The motives for undertaking the LCA will determine the scope and substance of the decisions based on the LCA results (product development, policy development and implementation, accountability). It goes without saying that the potential scope of the LCA-based decision should constitute a major determining factor vis-à-vis the intensity of participation in the decision-making process by the various stakeholders.

Interests

Finally, it should be stressed that LCA-based decision-making may involve a substantial variety of domestic as well as foreign players with commercial, public or ‘idealistic’ interests. These may be the interests of the companies or government agencies commissioning the LCA (henceforth ‘commissioners’), third-party interests or the international legal and economic context. The decision-making process should therefore be designed such that none of the relevant parties can be overlooked, to avoid implementation of inappropriate environmental measures, assertion of improper environmental claims and debatable competitive advantages.

Based on our own experience in a variety of practical contexts and on information on other cases, we have undertaken a careful review of LCA-based decision-making, giving particular attention to the procedural embedding of LCA projects and subsequent processes. In doing so, we have focused on the kinds of situations arising in both the public and the corporate decision-making context. We have found that LCA-based decision-making is often embedded in a standing procedural framework, statutory or otherwise. In a number of cases this was a broader procedural context explicitly allowing for possible execution of LCAs (Dutch Voluntary Agreement on Packaging, ‘Eco-label’). In other instances, such as LCAs performed as part of Environmental Impact Assessment (EIA) procedures, this was not the case.

In this report we do not reflect on the diversity of practical cases as such, but focus directly on the results of that reflection process. Against this background, the following sections present a number of general insights regarding those process aspects that have proved most relevant.

With regard to the procedural embedding of LCA studies, we here distinguish a total of 15 dominant process characteristics (items), divided over four process aspects. For each process item one or more potential bottlenecks are identified, based on experience and theoretical understanding. Due attention should be given to these bottlenecks, by anticipating the problems that are likely to arise.

I. Implementation of LCA tasks

process item 1 Overall process management
process item 2 Role and duties of the process manager
process item 3 Role and duties of the LCA practitioner
process item 4 Role and duties of the reviewer(s)

II. Representation of interests

process item 5 Selection of process participants
process item 6 Structures and interests to be respected
process item 7 Participation
process item 8 Commitment

III. Process progress

process item 9 Balance between content and process
process item 10 Balance between speed and thoroughness
process item 11 Outside influences
process item 12 Honouring input

IV. Process outcome

process item 13 Clarity of purpose
process item 14 Relation between LCA findings and process conclusions
process item 15 Relation between process conclusions and implementation

1.3.4 Implementation of LCA tasks

Satisfactory execution of the numerous activities involved in an LCA project requires a clearly delineated assignment of tasks among the various parties and tight overall management of responsibilities.

Process item 1: Overall process management

In most cases an LCA will not be executed by the initiating party but by one or more specialists, commissioned by one or more stakeholders. This may take a number of forms. The simplest configuration involves a single person representing a company or organisation - with a clearly defined purpose - and fulfilling the role of commissioner as well as overall process manager. If there are several parties involved and it is deemed desirable to give the various parties a fair chance of making an optimum contribution and the contracting (i.e. paying) party or parties are also willing to provide that opportunity, the role of 'commissioner' may be shouldered jointly by the full compliment of stakeholders. In such cases responsibility for overall project management is assigned to an independent process manager. In practice, of course, there will be numerous variations between these two extremes.

Experience shows that overall progress of the LCA process may be hampered by several factors:
- the commissioning party or parties have insufficient LCA know-how prior to their formulating the LCA assignment;
- the supervisory committees set up by the commissioners have an inadequate decision-making framework; this is particularly relevant when the commissioners have conflicting interests;

– financial responsibility for the LCA contract is undivorced from responsibility for overall project management.

Process item 2: Role and duties of the process manager

To ensure optimum project progress and stakeholder involvement a process manager may be appointed. A process manager may in principle fulfil any number of duties, with a strong emphasis on substance or purely process-oriented, on behalf of a commissioning party or entirely independent, purely as a mediator or simultaneously managerial and directive. If there is ambiguity in this area or if roles and duties are interpreted differently by various parties to the process, the status of the process manager may suffer and with it the status of the process itself.

Experience shows that overall progress of the LCA process may be hampered by several factors:
– the role and duties of the process manager are not clearly delineated prior to the start of the process;
– inadequate attention is paid to communicating the process manager's specific role and duties to all process participants;
– there is insufficient scope for assessing the process manager's performance in the various stages of the process.

Process item 3: Role and duties of the LCA practitioner

The task of the person(s) and/or organisation executing the LCA (the 'LCA practitioner') is to perform all the activities necessary to deliver the end product agreed to in the relevant contract. Generally speaking, this end product will take the form of an LCA quantifying the environmental profile of one or more product systems. It has been found in practice that an LCA often takes longer to perform than schedules permit. Interim revision of the contract is not unusual, particularly in the case of LCAs for use in comparative, public reviews.
A well-conducted LCA does not automatically lead to a good process result (through lack of consensus or commitment, for example). Conversely, an LCA deficient in substance may be blamed - rightly or wrongly - on a problematic process. On this playing field a conflict may arise between the interests of the party executing the LCA and the best interests of the process.
Another possible area of tension concerns the relation between the LCA practitioner on the one hand and the commissioning party and other stakeholders on the other. In the case of LCA-related conflict among stakeholders it may be the task of the LCA practitioner to expose fallacious thinking, signal inconsistencies and employ sensitivity analysis to quantify the effects of different assumptions, for example. In situations like these due care should be taken, however, to ensure that the LCA practitioner shows no 'bias'. One thing that must be avoided, for instance, is that the interests of the commissioning party lead to the product systems under study being modeled to respect considerations of corporate strategy rather than the system requirements specified. (In an assessment of possible future strategies, disagreeable product systems could conceivably be torpedoed through adoption of unfavourable assumptions regarding service life (short), resource consumption (high, polluting) and transport and cleaning processes (polluting).) In such cases the LCA practitioner can all too easily become party to a conflict regarding system boundaries and assumptions.

Experience shows that overall progress of the LCA process may be hampered by several factors:
– the LCA practitioner has not reflected sufficiently on the admissibility of the project terms and constraints as specified by the commissioning party;
– there is insufficient, diffuse regard for the role and responsibilities of the LCA practitioner in terms of both the LCA results and the overall process result;
– the LCA practitioner is insufficiently perceptive of the interests at stake and the conscientiousness this role demands;
– the LCA practitioner adopts an insufficiently independent stance, thus compromising the objectivity of the process.

Process item 4: Role and duties of the reviewer(s)

The aim of a (critical) review is to verify that an LCA study satisfies the set criteria with respect to such issues as methodology, system modeling, data selection and reporting mode. If LCA-based decision-making is to be worthwhile, it is of the greatest importance that one or more rigorous reviews be carried out. This is illustrated by the inclusion in ISO 14040 of background considerations and process criteria with regard to such reviews.

Experience shows that overall progress of the LCA process may be hampered by several factors:
- the independent status of the critical review and/or panel review within the decision-making process is not guaranteed;
- review duties are undermined because the status of the review(s) in the overall process is unclear (in terms of planning, for example);
- there is no explicit distinction between the advisory and evaluatory responsibilities of the critical reviewer(s) and/or panel;
- there are no clear arrangements regarding the relationship between critical reviewer and panel duties and reporting by these parties;
- there is insufficient clarity regarding the scope of the scheduled review reports and any constraints imposed on the review, either prior to or during the process.

1.3.5 Representation of interests

To ensure that decision-making is sufficiently transparent, it is essential that all relevant parties be involved in the decision-making process and that these parties can be confident that their interests are addressed as satisfactorily as possible within the adopted framework of the process.

Process item 5: Selection of process participants

It is of the utmost importance that the circle of participants engaged in the process be sufficiently wide. Although there may be an understandable urge to move forward, every endeavour should be made to avoid exclusion of relevant stakeholders and the loss of authoritativeness with which this is likely to be associated. In this context it should be borne in mind that those representing the parties involved must have sufficient 'commitment power'. Those being represented should consider themselves sufficiently bound to the outcome of the process.
Another point meriting due attention is balanced representation. 'Impassioned' parties for whom the stakes are high will send a relatively heavy delegation with plenty of time and know-how, while less impassioned parties will send lighter-weight representatives. This may upset the symmetry of the process and complicate overall progress.

Experience shows that overall progress of the LCA process may be hampered by several factors:
- inadequate prior consideration is given to the required breadth of representation, with no proper inventory of stakeholders;
- the stakeholder representatives recruited have insufficient commitment power;
- representation is unbalanced (inequality, asymmetry);
- process participants lack sufficient basic knowledge regarding LCA and the product systems under study; this is particularly important in the Goal and scope definition phase of the LCA.

Process item 6: Structures and interests to be respected

A process may sometimes be structured in such a way that the issues at stake coincide with the vital interests of (some of) the parties involved. Jeopardisation of these interests may then disrupt the entire process. Companies often state that they are willing to participate in a process only on condition that sensitive corporate information need not be made available to other parties (or only confidentially). The core interest of many public interest groups is to fulfil their role as moulders of opinion. Consequently,

they are not likely to take part in a process if this involves their not being allowed to publicise their positions in any way. Politicians, administrators and their representatives bear political responsibility and this is the key interest that they represent. LCA processes may not be designed such that this political accountability to representational bodies is compromised.

Experience shows that overall progress of the LCA process may be hampered by several factors:
- there is excessive discussion, prior to as well as during the process, about secrecy arrangements, through aversion to confidentiality on the one hand and insufficient explicit detailing and limitation of the scope of confidentiality on the other;
- prior to the process, there is insufficient detailing of how maximum freedom of action for all parties to the process is to be guaranteed, at the same time ensuring the least damage to the process as such;
- the primacy of the political domain is inadequately respected, while the commitment of political representatives is rendered insufficiently explicit;
- there are deficiencies in the organisational structure linking the process participants/representatives to their 'constituency'.

Process item 7: Participation

It is a well-known fact that at the outset of a process some parties take only a limited interest in the proceedings. The main reason is that it is not yet sufficiently clear to these parties what turn the process will take. Time and money must be invested in participation in a process whose outcome is still unclear. These actors therefore are insufficient motivated to participate in the process, even though they have every opportunity to influence the decision-making process. In the final stages of the process the picture is reversed. Certain actors will be particularly interested in continued participation, for the products of the process are becoming clearer all the time. At the same time the most important decisions will already have been taken, and the potential for influencing the process will be restricted accordingly. In short, there is major scope for influencing the decision-making process at its outset when participation is low, and far less scope for influence when participation is high.
Another phenomenon that deserves mention is the so-called 'participation paradox'. The aim of suitably broad-based participation is to improve the quality and support base of the decisions to be made. The paradox is that participation may have the opposite effect. Using the knowledge they have gained by participation in the process, parties are in a better position to oppose the decisions made once it is over. If they had acquiesced to a process outcome they were later to oppose, the party concerned might be accused of opportunistic conduct. For precisely this reason it might then be more appealing for the party to withdraw from the process before its termination.

Experience shows that overall progress of the LCA process may be hampered by several factors:
- there is a lack of explicit prior commitment by the process participants; there is insufficiently rigorous implementation of the agreed rules of play;
- there is insufficient encouragement of input from hesitant participants in the preliminary and initial phases of the project;
- some participants exert a disproportionate influence in the final project phase, thereby prompting undue reiteration of already finalised process elements;
- participants are replaced by heavier 'fresh' delegates and use made of 'fully functional' deputies.

Process item 8: Commitment

The best way to guarantee participants' commitment to the process is to ensure they have confidence in its outcome. To ensure participants feel sufficiently comfortable it is essential that their respective positions be duly protected. On the one hand, this has implications for process design: vital interests must be respected (cf. process item 6), while potential conflicts can be anticipated through wise design of the process structure. On the other hand, measures should be taken to limit the pressure exerted on parties during the process. In this respect it is important to provide due scope for postponing commitment wherever possible. It is not always wise to ask parties to commit themselves to all kinds of subsidiary decisions early in the process. Due allowance should be made for the complexity of LCAs:

matters that at first seem crucial may later transpire to be mere details, while details may later prove quintessential.
Ultimately, commitment to the final outcome of the LCA and the associated process is only possible on the basis of due insight and understanding. In practice this means that participants must undergo a learning process vis-à-vis LCA execution or the product systems being analysed, or both. Learning processes may be hampered if commitments are to be made at an early stage of the proceedings. In this respect it should always be remembered that the LCA process can only serve as a learning process for participants if the didactics are not overly disturbed by apprehension about unforeseen consequences.

Experience shows that overall progress of the LCA process may be hampered by several factors:
- participants' knowledge and understanding on LCA execution and the product systems under analysis consistently lags behind;
- participants are insufficiently motivated to develop a consistent picture in terms of content;
- 'binding' decisions are scheduled too early in the process.

1.3.6 Progress of the decision-making process

A process geared towards LCA-based decision-making can take a variety of different forms and the rate of progress will be determined in no small measure by how this process is designed.

Process item 9: Balance between content and process

The balance between content and process may be upset in two ways:
1. 'Process drives out content': parties may be so focused on consensus that issues of substance are insufficiently addressed.
2. 'Content drives out process': parties are overconcerned with details of substance that are irrelevant for achieving consensus.

Experience shows that overall progress of the LCA process may be hampered by several factors:
- there is too much focus on consensus and too little in-depth treatment of substance;
- there is too much focus on content, although this has little or no relevance for the final process outcome;
- participants shy away from real differences and conflicts and lose themselves in detail;
- participants 'take refuge' from the content in the process, because they are insufficiently knowledgeable at the outset of the process, for example;
- participants 'take refuge' from the process in the content, embarking on unnecessarily detailed descriptions to gain time, for example.

Process item 10: Balance between speed and thoroughness

Much criticism of the process approach is grounded in the view that processes are slow and cost too much time to work through. Due consultation of all the relevant stakeholders is a time-consuming process. As the process moves forward, parties moreover often learn a lot about their own conceptions and views, which become less secure as a result. This, too, may in turn affect the speed of the decision-making process. It is also true that LCA-based decision-making processes have often overshot their schedules, owing either to insufficient experience with LCA or underestimation of the complexity of the product systems and interests at stake.

There is not always a willingness to accept 'lost time' for the sake of better-quality decision-making. Time considerations may, for example, tempt participants to focus early on in the process on certain issues or certain solutions. One consequence of this may be inadequate coverage of the full range of issues and potential solutions (cf. process item 13). This may threaten the authoritativeness of the process, certain parties claiming that particular options were not addressed during the process, thus rendering the outcome of the process debatable a priori.

Experience shows that overall progress of the LCA process may be hampered by several factors:
- the desired quality of the LCA is not properly established at the outset, in terms of either content or process, nor is proper attention given to the possible implications of this for the project duration (this does not exclude the possibility of participants learning together during the process and subsequently reviewing quality criteria and schedules);
- there are deficiencies regarding one or more of the following points:
  - adequate preliminary across-the-board consideration of relevant issues by all participants;
  - collective agreement on and elaboration of the approach to be followed;
  - appropriate detailing of process planning;
  - ditto monitoring of progress;

  (this form of process design should not be too rigidly interpreted)
- during the process there is too little reflective evaluation vis-à-vis speed and thoroughness;
- there is inadequate scope for postponing the decision(s) if new insights arise or parties learn from the process;
- there is insufficient consultation about and explicit reformulation of de facto revisions regarding quality criteria and/or completion date.

Process item 11: Outside influences

Even if the circle of participants has been cast sufficiently wide - against the backdrop of the process goal - there are often (external) parties with a different kind of interest in the outcome of the process. The process may then be affected by outside influences, i.e. influences from the wider process context. The outcome of an LCA project may, after all, extend beyond the environmental profile of a given product system. A new contribution may have been made to methodology, for example, there may be improved understanding of the (market) operation of a product system and/or product chain, or a new basis of comparison for other process results, etc. For the outside world, as well as for certain process participants, it may be very important to give these aspects of the project specific status as subsidiary goals. All this means additional risks as well as opportunities with respect to the success of the project.

Experience shows that overall progress of the LCA process may be hampered by several factors:
- insufficient use is made of the advantages arising from the fact that process participants and outsiders recognise subsidiary process goals of greater or lesser importance;
- insufficient action is taken to combat the drawbacks arising from the fact that process participants and outsiders recognise subsidiary process goals of greater or lesser importance.

Process item 12: Honouring input

In principle, LCA-based decision-making should run its course as an open and transparent process. All the parties involved must be able to contribute their views and see them sufficiently honoured. Some parties may also consider the process as a kind of trap, however: by participating in the process they become committed to its outcome, without having been able to make any contribution of substance. Perceptions of this nature will be manifested differently by each of the various parties.
A different or additional problem may occur if there is too great a discrepancy between the actual process results and the actual value of the LCA results, on the one hand, and the impressions gained by the world outside. If some parties see their views reasonably well honoured but are meanwhile confronted with exaggerated 'image-building' efforts by other parties, they may be prompted to withdraw. This is particularly relevant when public relations departments gloss over or disregard participants' comments on the value of the LCA results and/or process results.

Experience shows that overall progress of the LCA process may be hampered by several factors:
- there is unfounded 'image-building' based on provisional/anticipated LCA and process results;
- there is insufficient respect for the wishes of some parties that announcement of provisional/anticipated LCA and process results should be accompanied by these parties' comments;

– there is no prior establishment of one or more opportunities for interim evaluation during which participants have an option to withdraw (giving their reasons for doing so).

1.3.7 Process outcome

In the LCA-based decision-making context a distinction should be made between an LCA result, the process conclusion and the process result. An LCA result is the sum total of results yielded by a life cycle analysis. The process conclusion is the conclusion drawn about a problem identified prior to the start of the project, motivated by an LCA result, or after comparison of one or more LCA results. The process result is the result the process conclusion being implemented.
This section is concerned with the steps followed in moving from goal to LCA result, process conclusion and implementation.

Process item 13: Clarity of purpose

It is crucially important that the goal of an LCA study be formulated such that the ultimate results of the LCA are optimally suited for use in the framework of the process goal. In the first place, this means that before the LCA study is started there should be measured reflection on the potential significance of the LCA results within the established framework of system boundaries and assumptions vis-à-vis the product system. Second, it means that the LCA study must be carried out as an iterative process, with ongoing reflection throughout. It should be emphasised that considerable prior attention should be devoted to formulating a clear-cut goal for both the process result and the LCA result. It is better to define the product system as appropriately as possible at the outset than to patch up a deficient definition on the basis of supplementary sensitivity analyses and considered options for subsequent improvement. A misfit between the final LCA result and the intended process result is a waste of research time and process time, moreover.

Experience shows that overall progress of the LCA process may be hampered by several factors:
– at the beginning of the LCA there is no critical, collective reflection on the goals of the commissioning party or of the LCA (with respect to adequacy and feasibility) by all the parties involved;
– during the process a lack of exchange between the LCA commissioner and practitioner prevents tightening or revising of established goals from being discussed where necessary.

Process item 14: Relation between LCA findings and process conclusions

The principal result of an LCA, or a series of comparative LCAs, is one or more environmental profiles: a review of the environmental scores of one or several product systems. LCA-based decision-making involves interpreting these profiles and drawing appropriate conclusions. This interpretative step has several aspects: the value the decision-makers attach to the environmental scores as such, the margins of error to be considered in doing so, insights concerning possible options for improvement, the relative importance of the environmental themes considered, how the modeled product system conforms to pictures of reality, and so on. Ultimate process conclusions often prove to be based not only on LCAs, but also on considerations of corporate economics or general policy concerns, for example.
It should be borne in mind that although both the interpretative step and the last-mentioned step are more difficult to objectify than the actual LCA process, it is feasible to design and implement appropriate procedural arrangements.

Experience shows that overall progress of the LCA process may be hampered by several factors:
– prior to the LCA process, there is no established decision-making procedure indicating as explicitly as possible how to deal with the issue of balancing considerations stemming from the LCA relative to considerations of corporate economics, policy and so on;
– insufficient prior consideration is given to procedures for handling the following issues in the conclusion phase:
– the margins of error in the environmental scores;

- the respective weighting of the environmental themes considered;
- insights obtained after considering options for improvement;
- insights obtained after performing sensitivity analyses (vis-à-vis assumptions in the modeled product systems, for example).

Process item 15: Relation between process conclusions and implementation

A key question in many decision-making contexts is how binding the process conclusion is to be. Are all the conclusions to be implemented exactly as they stand? Or is there to be some leeway, with the party or parties implementing the conclusions operating under certain degrees of freedom? The advantage of the first option is that the conclusions bear major significance; parties can count on their findings being implemented. The drawback of a binding process conclusion is that it can loom as a threat over the conclusion process (and the preceding analysis), for if participants are aware that the process conclusions are to be implemented as they stand, there will be a strong incentive for certain parties to resist particular conclusions.

Experience shows that overall progress of the LCA process may be hampered by several factors:
- the conclusions drawn during the process and the ultimate process conclusions are too binding with respect to subsequent measures;
- the opposite is true, permitting opportunistic use and abuse during the process;
- the process is insufficiently transparent, compounding these opportunities for process abuse;
- no measures are taken to avoid opportunistic use and abuse of non-binding conclusions after termination of the process.

1.4 Stepwise structure for Environmental Life-Cycle Assessment

As already mentioned in Chapter 1, this Guide adheres rigidly to the four main phases of LCA specified by ISO (see). In this Guide we elaborate a logical, stepwise structure for each of these four phases, based on general practical experience of the logical sequence of actions undertaken in the course of an LCA study. In naming the steps we have taken the ISO phraseology[1] as our point of departure wherever possible. No iterations are specified in the Guide, for in principle the outcome of any step may lead to a revision of previous steps. In practice, iteration between steps will be so frequent and manifest itself in so many different ways that it would be impossible to elaborate them all in a guide. In this sense, the described steps specify a logically ordered structure that allows for any degree of iteration desired. Procedural aspects with broader ramifications are specified for each phase separately.

1.4.1 Goal and scope definition

ISO 14041 (1998E) breaks down the Goal and scope definition phase of an LCA into the following steps or elements (clause 5, p.4):
- Goal of the study;
- Scope of the study;
- Function, functional unit and reference flow;
- Initial system boundaries;
- Description of data categories;
- Criteria for initial inclusion of inputs and outputs;
- Data quality requirements;
- Critical review;
- Study report.

[1] This does not imply that the authors consider ISO phraseology to be the most appropriate in all cases. However, deviation from the ISO wording would probably not be helpful in achieving an understandable and widely acceptable Guide.

Based on a review of steps and items proposed for the Goal and scope definition phase by ISO 14040 (1997E) ISO 14041 (1998E), Heijungs *et al*. (1992), Lindfors *et al*. (1995a) and Wenzel *et al*. (1997), we propose to distinguish the following steps for Goal and scope definition:

- **[Procedures];**[1]
- **Goal definition;**
- **Scope definition;**
- **Function, functional unit, alternatives and reference flows.**

Although the structure is broadly similar to ISO 14041 (1998E), there are several differences and additions:

1. Procedures covers the ISO element 'Critical review', but is a more general term indicating the possible usefulness of covering other procedural aspects like stakeholder participation in a particular LCA study. In the present Guide guidelines on this topic are given for each individual LCA phase and the Procedures step therefore also recurs in the Inventory analysis, Impact assessment and Interpretation phases. However, in Part 3 - Scientific Background (this document) - Procedures are not discussed for each step separately but in a single, comprehensive section of this chapter (Section 1.3). This is because 'Procedures' is a relatively new topic and it is not yet particularly useful to discuss developments since 1992 for each LCA phase individually. A separate background study has been devoted to the topic (De Bruijn & Van Duin, 1998; Van Duin & De Bruijn, 1998).
2. To the ISO step 'Function, functional unit and reference flow' the choice of alternatives has been added in order to emphasise that it is here that selection of (product) systems for comparison takes place, if relevant in a given LCA study.
3. Initial system boundaries, description of data categories, criteria for inclusion of inputs and outputs and data quality requirements now constitute elements of the Inventory analysis.
4. In each step distinct guidelines are given for reporting and for issues for Interpretation. Rather than devoting specific sections to these two items, which would lead to considerable double-wording, we have deemed it more useful to discuss them in context as they arose. The term 'issues for Interpretation' has been chosen to indicate that this step is related to the Interpretation phase. Identification of issues for Interpretation helps create a checklist of all Goal and scope, Inventory and Impact assessment issues of possible relevance for the Interpretation phase

The items procedures, reporting and issues for Interpretation are treated in the Inventory analysis and the Impact assessment phases, in the same way as described above for Goal and scope definition. Procedures and reporting are treated similarly in the Interpretation phase, too, but for obvious reasons issues for Interpretation is then a superfluous item.

One remaining point is what the Scope definition step exactly comprises. It is not clear from the ISO standards what should be stated about the issues and/or steps in the Scope definition and what in the Inventory analysis and Impact assessment.

The suggestions provided by ISO 14040 and 14041 on Scope definition are slightly different. Under the heading 'scope of the study' a list of items is given in ISO 14040 (1997E), clause 5.1.2 (and referred to in ISO 14041, clause 5.3.1), which shall be considered and should be clearly described:

- the function of the system;
- the functional unit;
- the system to be studied;
- the system boundaries;
- allocation procedures;
- the types of impact and the methodology of Impact assessment and subsequent Interpretation to be used;
- data requirements;
- assumptions;
- limitations;
- the initial data quality requirements;
- the type of critical review, if any; and
- the type and format of the report required for the study.

We suggest that these elements of the ISO Scope definition be split into three clusters. The first of these covers a number of basic choices for the entire study, including:

[1] "Procedures" is a separate step in Part 2a - Guide - but not in Part 3 - Scientific background.

– temporal coverage (including infinite time horizon in modeling economic flows, environmental interventions and impacts);
– geographical coverage;
– Technology coverage;
– coverage of economic processes (initial system boundaries);
– coverage of environmental interventions and impacts;
– mode of analysis[1]; and
– level of sophistication.

In this Guide these issues will be elaborated in the step 'Scope of the study'.

The second cluster covers basic choices regarding function, functional unit, alternatives and reference flows. Here, a separate step of Goal and scope definition is devoted to these key issues of an LCA study.

The third cluster comprises critical review and reporting, viz. of all the basic choices with respect to functional unit, systems compared, Inventory analysis, Impact assessment and Interpretation. This so-called Goal and Scope Report is drafted for the purpose of critical review and stakeholder comments and, in comparison to ISO 14040 and 14041, is here extended to cover reporting on all the main choices of all the methodological steps distinguished. For this reporting dimension, separate guidelines are given in each step of the Goal and scope definition.

The relation between the steps of Goal and scope definition in this Guide and those derived from ISO 14041 is shown in Figure 1.4.1.1.

[1] "Mode of analysis" refers to a subject that is not (yet) treated in the ISO document but has become a key topic in the LCA debate. It concerns the distinction between descriptive and change-oriented applications, also known as the "marginal-average discussion" (*cf.* Udo de Haes and Wrisberg, 1997; Frischknecht, 1998). The distinction between these two types of application appears to be extremely important, for two reasons:

1. it enables an explicit connection to be made with the application and hence with the Goal definition, the functional unit, etc.;
2. it may have major consequences for methodological details during Inventory analysis, Impact assessment and Interpretation.

As indicated above, this Guide focuses on change-oriented analysis for long-term structural decisions.

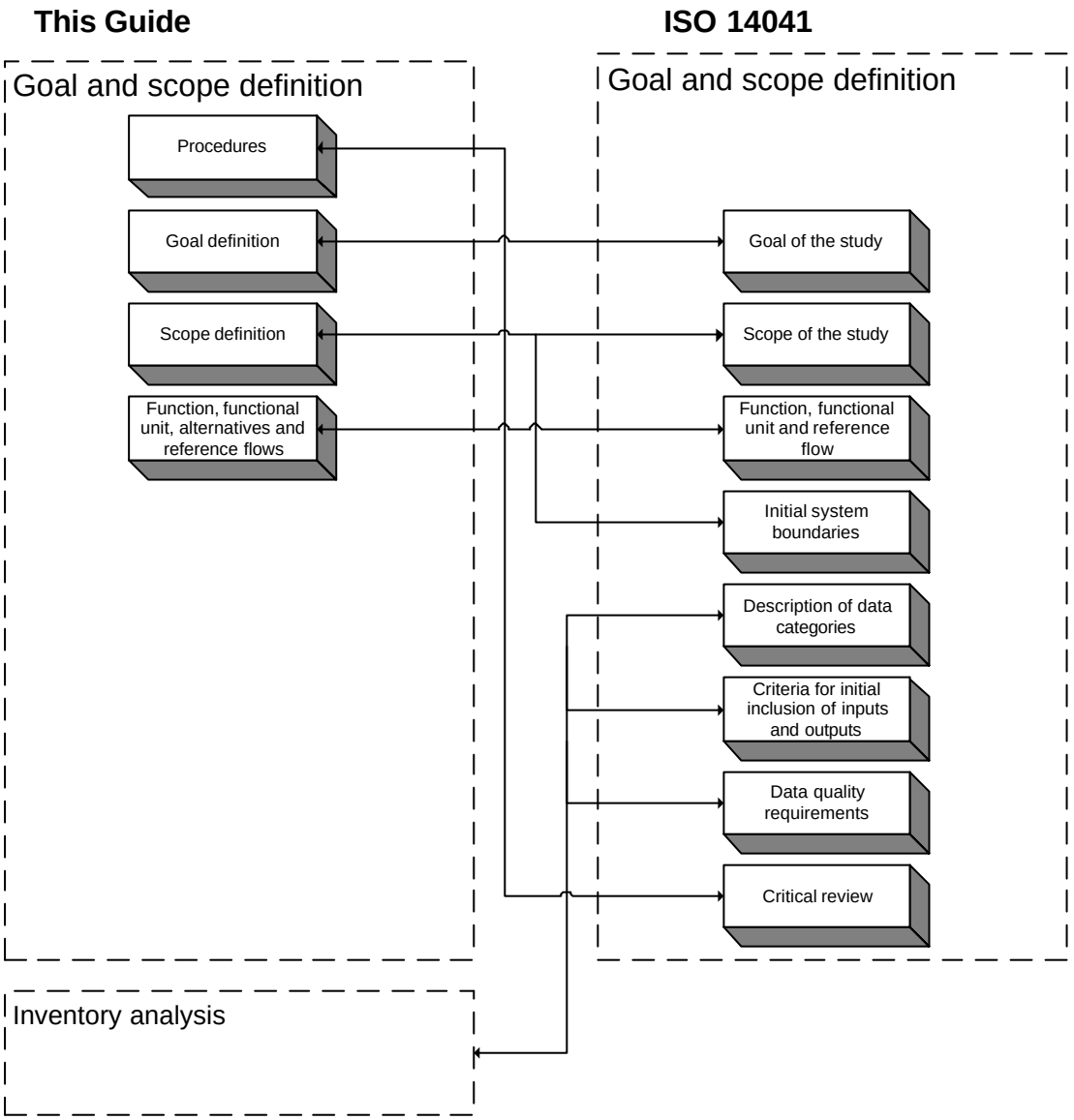


Figure 1.4.1.1: The relation between the Goal and scope definition steps distinguished in this Guide and in ISO 14041 (for reasons of presentation the sequence of ISO steps has been adapted).

1.4.2 Inventory analysis

In ISO 14041 (1998E) the (Life Cycle) Inventory analysis phase ('LCI') is broken down into the following operational steps (ISO 14041, clause 6.1, p.8):

- Preparing for data collection;
- Data collection;
- Validation of data;
- Relating data to unit process;
- Relating data to functional unit;
- Allocation and recycling;
- Data aggregation;
- Refining the system boundaries.

In addition to these operational steps the topics 'limitation of LCI (interpreting LCI results)' and 'study report' are mentioned in ISO 14041 (1998E).

Based on ISO 14041 (1998E) and taking into account the steps distinguished in Heijungs *et al*. (1992), Lindfors *et al*. (1995a), Curran (1996) and Wenzel *et al*. (1997), in this Guide we distinguish the following steps of the Inventory analysis phase:

- **[Procedures];** [1]
- **Economy-environment system boundary;**
- **Flow diagram** (one item of 'Preparing for data collection')**;**
- **Format and data categories** (ISO step 'Data categories')**;**
- **Data quality;**
- **Data collection and relating data to unit processes** (including 'Relating data to unit processes')**;**
- **Data validation;**
- **Cut-off and data estimation** (= 'Initial system boundaries' and 'Criteria for initial inclusion of inputs and outputs'; in ISO 14041 these steps are part of 'Goal and scope definition');
- **Multifunctionality and allocation** (= 'Allocation and recycling');
- **Calculation method** (= 'Relating data to functional unit' and 'Data aggregation').

The steps listed cover all ISO steps and elements, but they are here grouped and sequenced differently under slightly different headings. There are a number of reasons for these additions to and deviations from ISO:

1. As the overall aim of this document is to provide a practical guide for LCA practitioners based on the ISO Standards, it is important to parallel the actual steps of a real-world LCA process as closely as possible.
2. The procedural aspects of each LCA phase are treated in a separate 'Procedures' section in the description of each phase, to avoid cluttering up the technical steps.
3. The ISO step 'Preparing for data collection' has been split into 'Flow diagram' and 'Data collection and relating data to unit processes'. The description of each unit process, units of measurement, data retrieval methods, etc. are all part of 'Data collection and relating data to unit processes'.
4. The ISO step 'Relating data to unit processes' is now part of the 'Data collection and relating data to unit processes' step, as 'Relating data to unit processes' has too little substance to be treated as an individual step and is inextricably linked with the activity of data collection.
5. 'Format' is lacking as a topic in the ISO framework and has been added here as part of the first data step: 'Format and data categories'.
6. Definition of the economy-environment system boundary is a step that is lacking in the ISO framework and it has therefore been added here.
7. All calculation steps have been brought together under 'Calculation method', but with a distinction still being made between non-aggregated and aggregated inventory results.
8. Cut-offs are usually introduced because of data deficiency and general time and resource constraints on data retrieval. It is only after data collection that this problem becomes evident and it has therefore been put after data collection. As cut-off may often be avoided by data estimaton techniques such as input-output analysis (see Section 3.8), the name of this step has been extended to 'Cut-off and data estimation'.
9. The step 'Refining the system boundaries' is related to the iterative character of LCA execution and, as argued above, is not elaborated as a separate step here.
10. Issues for Interpretation, reporting and procedures (including critical review) are treated in the same way as in the Goal and scope definition (see Section 1.4). The ISO topics 'Limitation of LCI (interpreting LCI results)' and 'Study report' are not treated in this Guide as separate steps but as part of the items 'Issues for Interpretation' and 'Reporting', respectively.

The relation between the steps distinguished in this guide and the steps of ISO 14041 (1998E) is shown in Figure 1.4.2.1.

[1] Parallel to the treatment of this subject in the Goal and scope phase, "Procedures" is a separate step in Part 2a - Guide - but not in Part 3 - Scientific background.

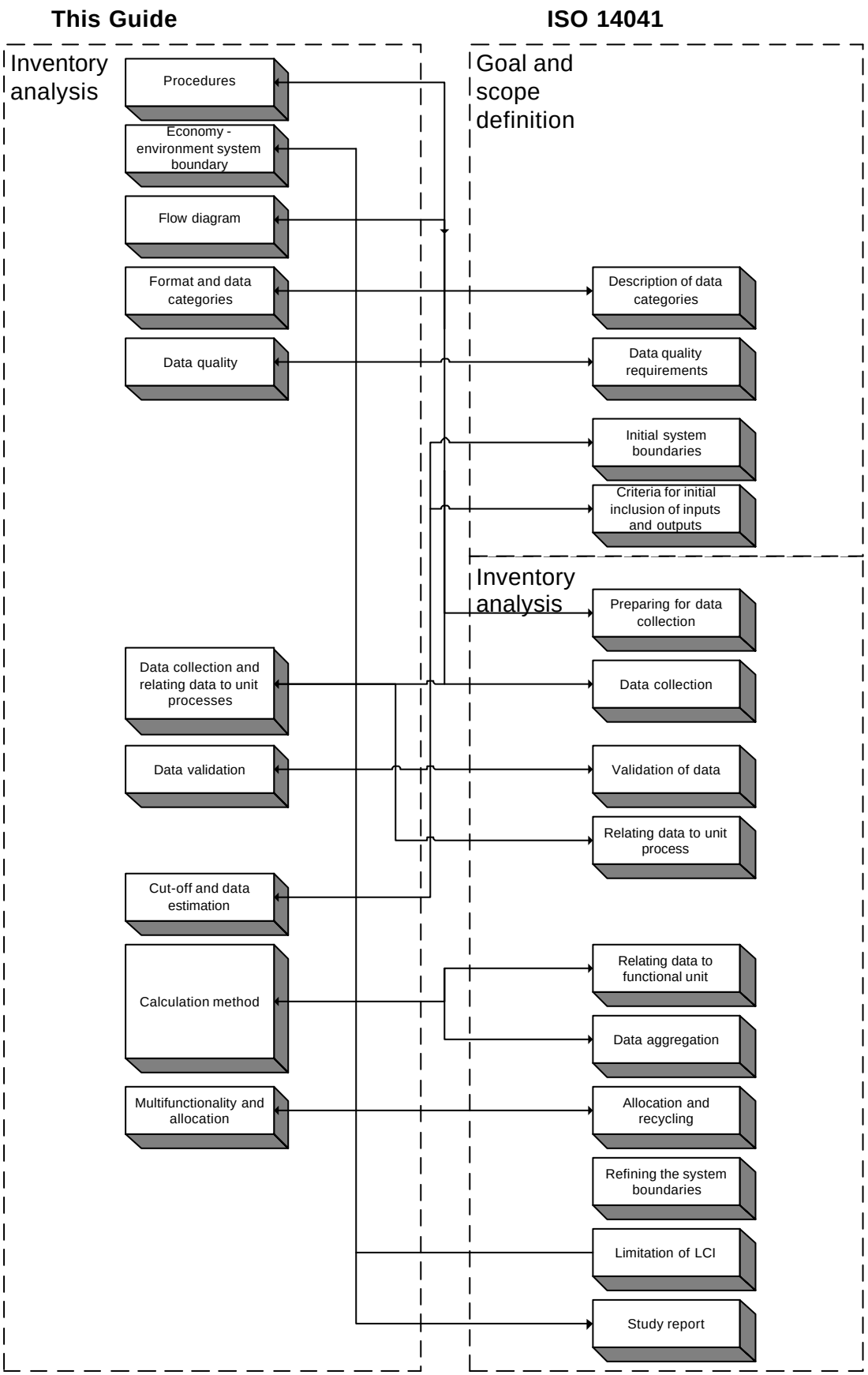


Figure 1.4.2.1: The relation between the Inventory analysis steps distinguished in this Guide and in ISO 14041 (for reasons of presentation the sequence of ISO steps has been adapted).

1.4.3 Impact assessment

ISO 14042 (2000E) distinguishes the following steps In the (Life Cycle) Impact assessment phase ('LCIA'):

- Selection of impact categories, category indicators and characterisation models;
- Assignment of LCI results (Classification);
- Calculation of category indicator results (Characterisation);
- Calculating the magnitude of the category indicator results relative to reference information (Normalisation);
- Grouping;
- Weighting;
- Data quality analysis;
- Limitations of LCIA;
- Comparative assertions disclosed to the public;
- Reporting and critical review.

The ISO framework is summarised in Figure 1.4.3.1.

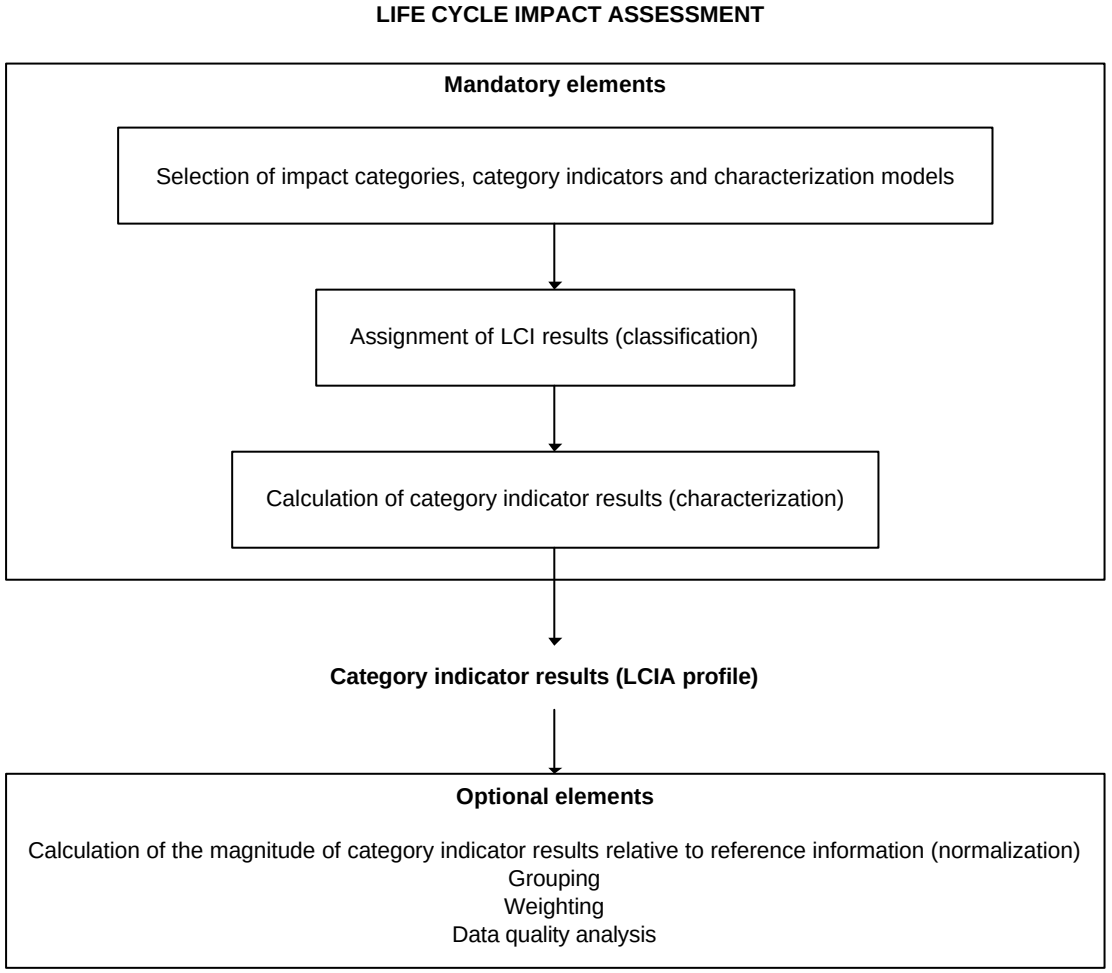


Figure 1.4.3.1: The ISO elements of the LCIA phase (source: ISO 14042, 2000E).

ISO 14042 distinguishes mandatory and optional elements. The first mandatory element is 'Selection and definition of impact categories, category indicators and characterisation models'. In this step impact categories (e.g. climate change) are identified along with appropriate category indicators (e.g. infrared radiative forcing) and the model used to derive the characterisation factors, i.e. the quantitative relationship between the interventions and the indicator, is established. The second mandatory element is 'Assignment of LCI results to category indicators (Classification)'. In the final mandatory element, 'Calculation of category indicator results (Characterisation)', guidance and requirements are provided for

calculating indicator results. Characterisation yields the ‘environmental profile’, consisting of a series of ‘indicator results’.
Besides the mandatory elements there are three optional elements. The first of these, ‘Normalisation’, covers calculation of the magnitude of category indicators relative to reference information. In the second optional element, ‘Grouping of indicator results’, impact categories are grouped into one or more sets involving a descriptive sorting or a prioritising ranking. The third optional element is ‘Weighting’, i.e. multiplication of indicator results or normalised results by numerical factors, with the aim of converting and possibly aggregating indicator results across impact categories into a single score or a small number of such scores. As a final optional element, ‘Data quality analysis’ may be performed to enhance understanding of the significance, uncertainty and sensitivity of the LCIA results.
The mandatory and optional steps described above imply that different trajectories can be adopted en route to the final LCIA result. These are illustrated in Figure 1.4.3.2.

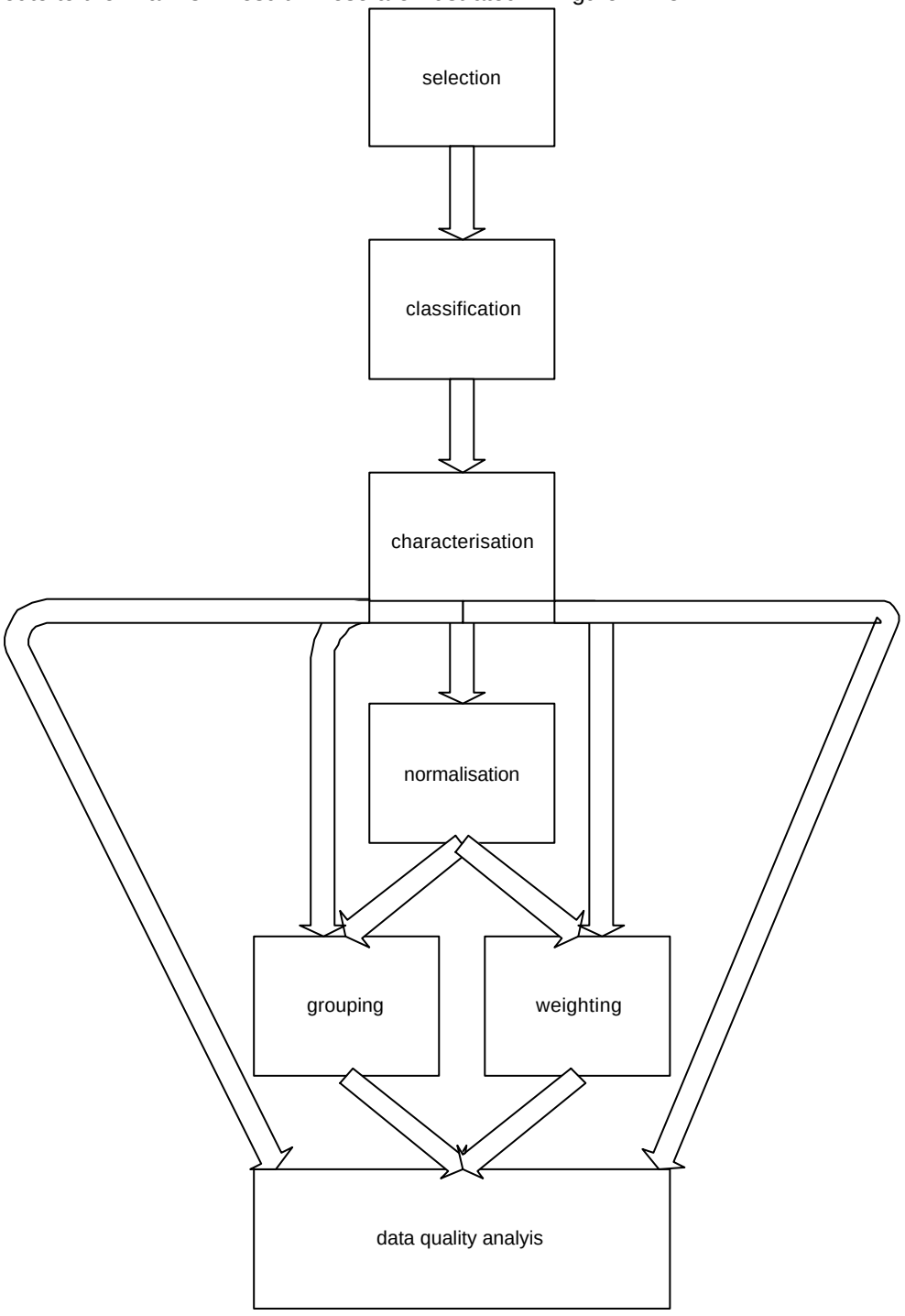


Figure 1.4.3.2: Options for combining mandatory and optional ISO LCIA steps.

Based on ISO 14042, in this new Guide we distinguish the following steps of the Impact assessment phase:

- **[Procedures];**[1]
- **Selection of impact categories;**
- **Selection of characterisation methods: category indicators, characterisation models and factors;**
- **Classification;**
- **Characterisation;**
- **Normalisation;**
- **Grouping;**
- **Weighting.**

This structure is broadly similar to ISO 14042 (2000E). The main difference is that 'Limitations of LCIA' and 'Comparative assertions disclosed to the public' are not discussed here as separate steps of the LCIA phase. The limitations of LCIA are treated in the general introduction to this Guide, under Goal and scope definition and Interpretation: in the former, because the general limitations of LCA (including those of LCIA) should be duly appreciated before conducting the actual LCA study and in the latter because they put the conclusions into due perspective. Comparative assertions are one possible application of LCA; this is determined in the Goal and scope definition phase and will steer several choices during the LCA study (including the choice of simplified versus detailed LCA, application of weighting methods, reporting guidelines, etc.). Issues for Interpretation (covering the ISO topic 'Data quality analysis'), reporting (covering ISO's 'Reporting') and procedures (covering ISO's 'Critical review') are treated in the same way as in Goal and scope definition (see Section 1.4).

The selection of impact categories, category indicators and characterisation models has been broken down into two steps, moreover, because these are clearly sequential elements.

Finally, this Guide does not entirely retain ISO's distinction between mandatory and optional steps. The Dutch LCA community feels that normalisation and data quality assessment (issues for Interpretation) constitute at least recommended, if not mandatory, steps and that these should be part of any LCA study.

The relation between the steps distinguished in this guide and the steps of ISO 14042 (2000E) is shown in Figure 1.4.3.3.

[1] Parallel to the treatment of this subject in the Goal and scope phase, "Procedures" is a separate step in Part 2a - Guide - but not in Part 3 - Scientific background.

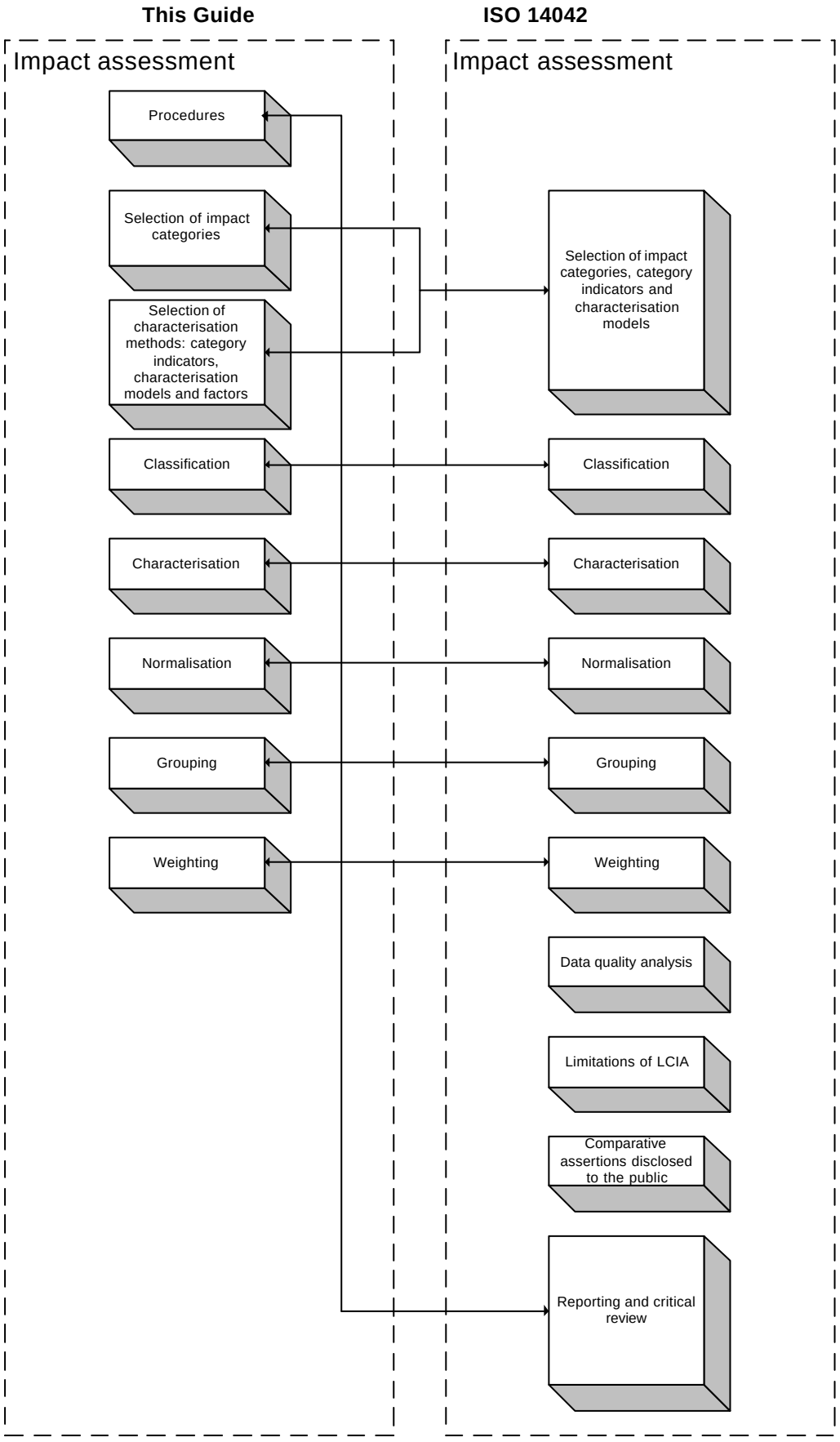


Figure 1.4.3.3: The relation between the Impact assessment steps distinguished in this Guide and in ISO 14042.

1.4.4 Interpretation

In ISO 14043 (2000E) the Interpretation phase is broken down into three elements:
- Identification of significant issues, based on the results of the LCI and LCIA phases of LCA;
- Evaluation, comprising completeness, sensitivity and consistency checks;
- Conclusions, recommendations and reporting.

Other elements of analysis, including a critical review, are considered additionally in clause 9 of ISO 14043.

Based on ISO 14043, in this Guide we distinguish the following steps of the Interpretation phase:
- [**Procedures** (Chapter 9 of ISO 14043)][1];
- **Evaluation of results:**
  consistency check
  completeness check
- **Analysis of results:**
  contribution analysis
  perturbation analysis
  sensitivity and uncertainty analysis
- **Conclusion and recommendations.**

This structure is broadly similar to ISO 14043. The main difference is that 'Evaluation' has been split into two parts, one of which is placed before 'Identification of significant issues'. The reason for this change is that if there are large inconsistencies or errors in the data or if the data is very incomplete, all further Interpretation steps become futile. A minor difference is the use of the term *analysis* instead of *check* for the sensitivity and uncertainty analysis[2]. 'Identification of significant issues' is here operationalised in two different steps: the contribution analysis and the perturbation analysis. Reporting and procedures are treated in the same way as for the Goal and scope definition (see Section 1.4). For obvious reasons, and to prevent endless iteration, issues for Interpretation is no longer included as an item in the Interpretation phase itself.

The relation between the steps distinguished in this guide and the steps of ISO 14043 (2000E) is shown in Figure 1.4.4.1.
There is an close relationship between the steps of the Interpretation phase and the other phases of the LCA. On the one hand, there is input from the other phases, because these identify relevant issues for Interpretation. On the other hand, the iterative nature of the LCA process allows for, and sometimes even demands, making changes in prior phases, as when errors are found or results prove to be 'too' sensitive to particular, debatable data, model choices, etc. during Interpretation.

[1] Parallel to the treatment of this subject in the Goal and scope phase, "Procedures" is a separate step in Part 2a - Guidelines - but not in Part 3 - Scientific background.
[2] Here, the term *check* is used to emphasise the fact that data or model choices are being checked, while the term *analysis* is used for more complex assessments requiring dedicated tools and so on.

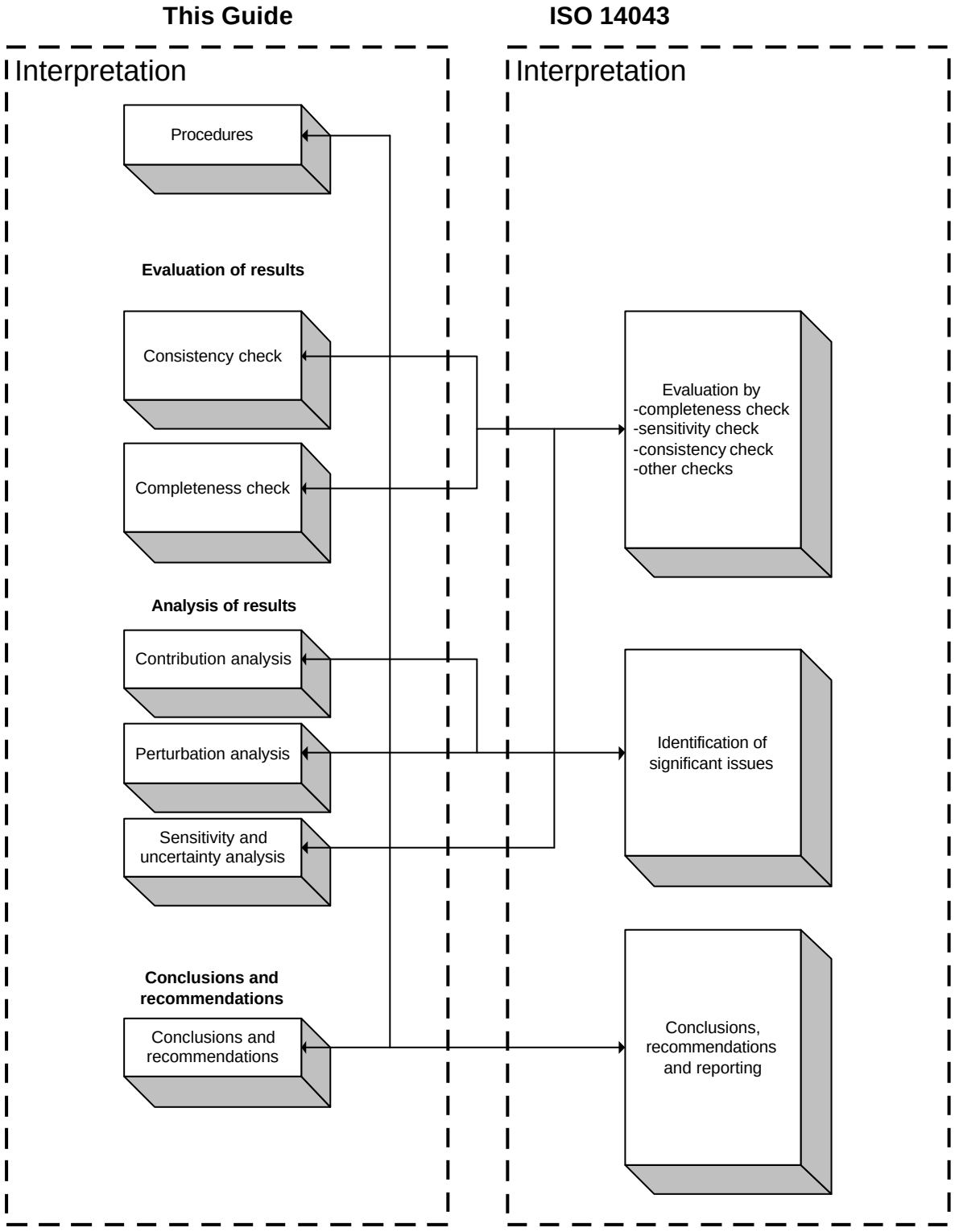


Figure 1.4.4.1: The relation between the Interpretation steps distinguished in this Guide and in ISO 14043.

1.5 Further reading guidance

In the following chapters the steps distinguished above are discussed for each phase of LCA, viz. Goal and scope definition, Inventory analysis, Impact assessment and Interpretation. Each step is discussed according to a fixed format: 'Topic', 'Developments in the last decade', 'Prospects', 'Conclusions' and 'Research recommendations'. However, as 'Procedures' is a fairly new topic in LCA with few

developments in the last decade to discuss, in this case this format is not particularly useful. The Procedures step is therefore not discussed in each phase individually, but as an integrated whole in Section 1.3. Another important issue to bear in mind while reading and using this Guide - and to stress once more at this point - is that it has become clear in the preceding introduction that there is no such thing as 'a correct LCA'. The sophistication of an LCA and the way in which it is elaborated depend on the specific situation in which LCA is being used as a decision support tool. In theory this might imply a practitioner having to develop custom-tailored methods for each LCA study anew. From a practical angle, however, this is obviously out of the question. Most choices vis-à-vis methods have broad ramifications, restricting the choices available in other respects. One cannot first opt for a steady-state model and subsequently opt for a change in capacity utilisation to supply the required inputs, as short-term and long-term perspectives would then be mixed. Even if one agrees on the main purpose of the LCA study, assuming that structural questions are to be answered there are still hundreds of choices to be made before a well-founded LCA model has been constructed. This is clearly beyond the capabilities of most practitioners and beyond the budget of most commissioners. To permit broad application of LCA some degree of standardisation is therefore essential.

The standardisation developed in this Guide goes beyond the ISO standards, as the latter specify only the general structure of LCA. True operationalisation involves so many choices that one can hardly expect the world to agree on all of them and formulate them in an ISO standard. The basic question is therefore: how to arrive at an operational LCA which at the same time providing the flexibility required for widely differing applications. We have endeavoured to steer a 'middle' course, on the one hand excluding certain applications and on the other specifying a number of standardised types of LCA and a framework within which deviations from these can be accomplished in a transparent manner.

The first restriction has already been explained: the prime focus of this Guide is to support structural decisions, using a long-term steady-state model for that purpose in the Inventory analysis phase and the most apt (but more diverse) models in the Impact assessment phase. This does not mean that short-term issues are deemed less important. It is merely our choice of focus, prompted by the aim of supporting long-term sustainability developments.

Within these limitations, a wide variety of possible methods still remains available. We elaborate two basic options in more or less systematic fashion.

- The first is a *detailed* LCA , which we believe to be representative for studies typically requiring between 20 and 200 days of work. The detailed LCA is the *baseline* LCA elaborated in this Guide.
- The second is a *simplified* version of LCA, typically requiring between 1 and 20 days of work. In this Guide we prescribe neither one option or the other, merely providing them as a ready reference within the overall framework of issues and steps already specified. Appropriate sensitivity analyses are also suggested. Practitioners wishing to deviate from the guidelines provided in this Guide for a specific step are at liberty to do so, but should clearly justify their decision accordingly. One may, for example, deviate from the economy-environment system boundary specified for detailed LCA, or choose a different time horizon for leaching of landfill. Such options are not specifically supported in this Guide, however, and their implementation is entirely the responsibility of the LCA practitioner. It seems advisable, though, to introduce such deviations in the form of a sensitivity analysis of detailed LCA, in order to retain reference to a more or less standardised type of LCA. Note that in a detailed LCA certain steps may be performed at the simplified level, and within a single step, one may indeed even opt to apply detailed guidelines for some unit processes or impact categories and simplified guidelines for others. Finally, note that a simplified LCA is not simple in the sense of being easy.
- Finally, on some topics an indication is provided of possible *extensions* for improving the quality of detailed LCA in those respects where shortcomings are most obvious. A key example is the absence of economic mechanisms in the LCA model, an unfortunate feature in cases where there are extreme inelasticities of supply or demand. In the case of rechargeable batteries, for example, a shift to non-cadmium types of battery will not result in a smaller influx of cadmium to the economy and hence will not lead in the long run to reduced cadmium emissions. This is because the supply of primary cadmium is extremely inelastic. In such a situation, LCA may yield a misleading outcome. For a number of situations possible extensions are therefore specified, both within an adapted LCA framework and as an addition to detailed LCA in Figure 1.5.1.

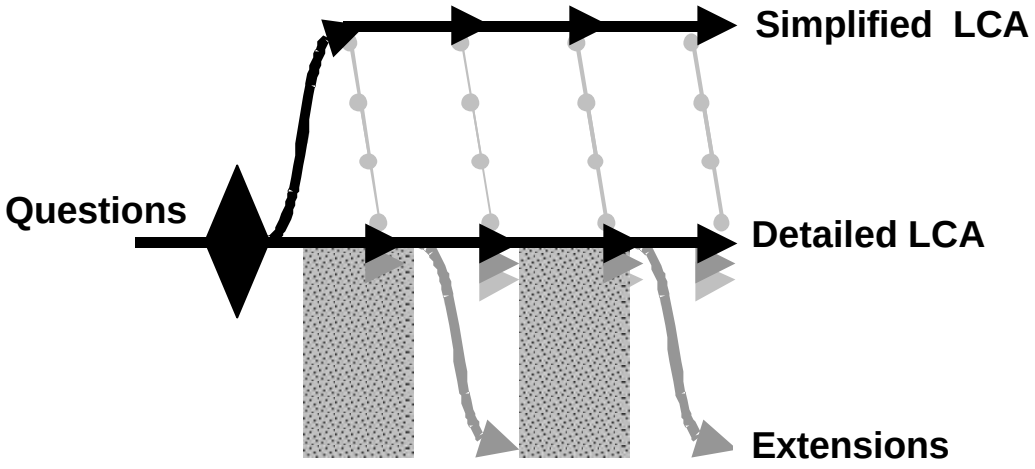


Figure 1.5.1: The relation between simplified LCA, detailed LCA and extensions.

In summary, simplified LCA and extensions build on detailed LCA, which is the baseline elaborated in this Guide.
The guidelines for the various steps of LCA - the scientific background to which is provided in the present Part - have not all been elaborated at the same operational level. The guidelines for one step may imply that in each and every LCA study a number of actions should be actively performed, while those for another step may imply that one need generally only follow the baseline proposals given in this Guide. In the 'Flow diagram' step, for example, a representative diagram of processes pertinent to the given LCA study will have to be drawn up in each and every study, while in the 'Selection of impact categories' step the baseline proposal (see Section 4.2) can generally be followed time and time again. In the latter case, one will of course have to evaluate whether the baseline proposal is sufficient, or whether other categories beyond the baseline proposal should be added for the specific LCA study in question.

Below we present a ready-reference overview of the methodological steps distinguished in this Guide for each phase of an LCA, with a reference to the section of the present Part dealing with each specific step (Table 1.5.1).

Table 1.5.1: Overview of methodological steps distinguished in this Guide, with reference to the section of this part dealing with that specific step.

Phase	Step	Section in Part 3	Main result
Goal and scope definiton	Procedures	1.3	Functional unit; alternatives compared
	Goal definition	2.2	
	Scope definition	2.3	
	Function, functional unit, alternatives and reference flows	2.4	
Inventory analysis	Procedures	1.3	Inventory table; other indications (e.g. missing flows)
	Economy - environment system boundary	3.2	
	Flow diagram	3.3	
	Format and data categories	3.4	
	Data quality	3.5	
	Data collection and relating data to unit processes	3.6	
	Data validation	3.7	
	Cut-off and data estimation	3.8	
	Multifunctionality and allocation	3.9	
	Calculation method	3.10	
Impact assessment	Procedures	1.3	environmental profile; Normalised environmental profile; Weighting profile
	Selection of impact categories	4.2	
	Selection of characterisation methods: category indicators, characterisation models	4.3	
	Classification	4.4	
	Characterisation	4.5	
	Normalisation	4.6	
	Grouping	4.7	
	Weighting	4.8	
Interpretation	Procedures	1.3	(Well-balanced) conclusions and recommendations
	Consistency check	5.2	
	Completeness check	5.3	
	Contribution analysis	5.4	
	Perturbation analysis	5.5	
	Sensitivity and uncertainty analysis	5.6	
	Conclusions and recommendations	5.7	

2. Goal and scope definition

2.1 Introduction

No explicit definition of the Goal and scope definition phase is given by ISO. A definition derived from ISO's work might run: The Goal and scope definition is the first phase of an LCA, stating the aim of an intended LCA study, the functional unit, the system alternatives considered, and the breadth and depth of the intended LCA study in relation to this aim (see also Figure 2.1.1).

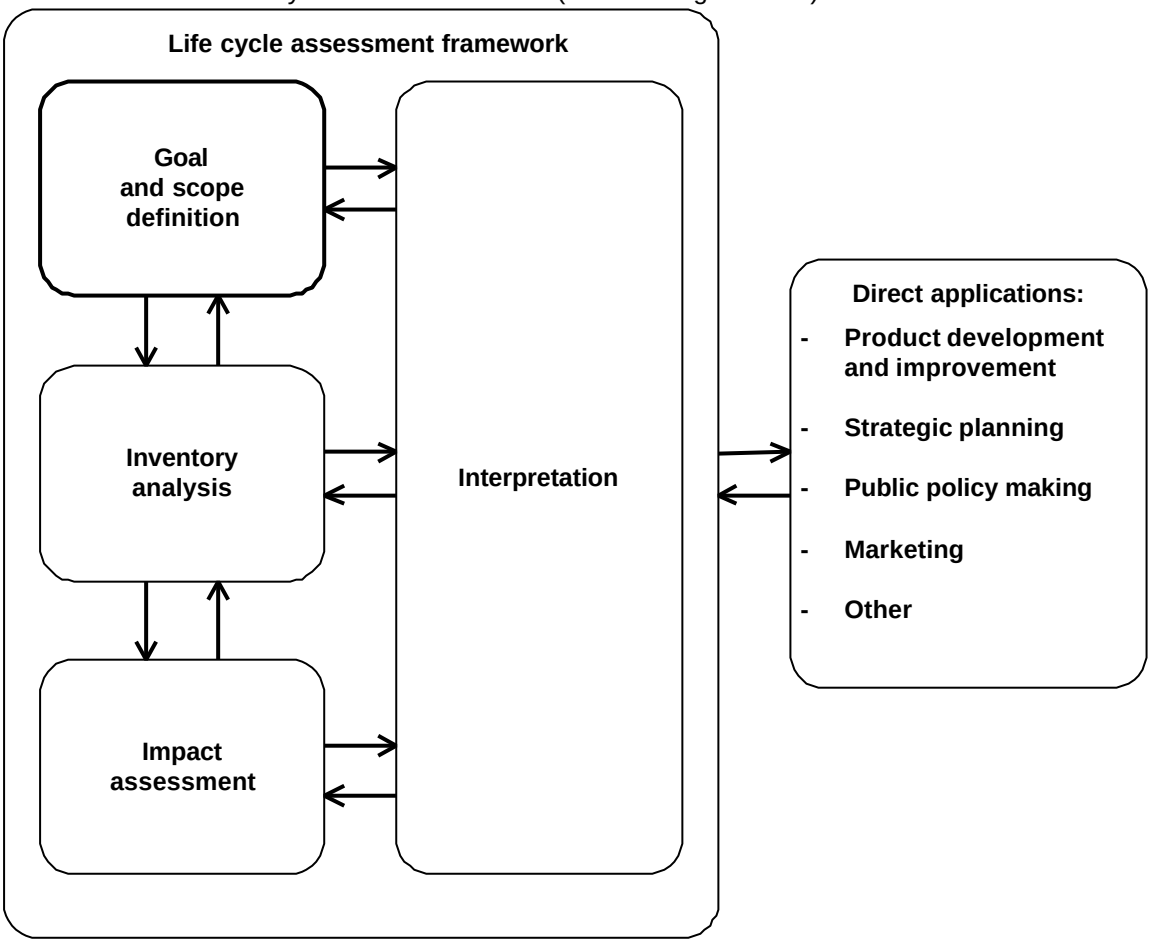


Figure 2.1.1: The Goal and Scope definition phase as part of the general methodological framework for LCA (source: ISO 14040, 1997E).

The Goal and scope definition anticipates the application, which might be to provide product information (e.g. by comparing product alternatives), 'public regulation' (e.g. product approval based on the results of comparison with a standard), product or process innovation (e.g. by identifying dominant processes in the environmental profile to obtain information about the potential effects of innovation) or as a tool for strategic studies based on policy scenarios. The scope of the study is also established at this stage, as a function of the time and money available and the intended application. Furthermore, the functional unit and the products to be investigated are defined. Finally, as in each phase (see Chapter 1), important issues relating to goal and scope are identified in and reporting guidelines established for each step (*cf.* Heijungs *et al.*, 1992).

The Goal and scope definition will largely be the result of discussions between the project commissioners, the practitioners and those with interests in the study results (interested parties or stakeholders). The procedural aspects of the goal and scope phase are therefore of particular importance.

The result of goal definition is an accurate description of the goal and scope of the study, the functional unit to be used and the (product) system(s) to be investigated.
Thus, the Goal and scope definition phase comprises four steps:
- Procedures (no special section in this Part; see Chapter 1);
- Goal definition (Section 2.2, p. 63);
- Scope definition (Section 2.3, p. 67);
- Function, functional unit, alternatives and reference flow (Section 2.4, p. 76).

Further points of departure for elaborating these Goal and scope definition steps are ISO documents 14040 (1997E) and 14041 (1998E) particularly with regard to the methodological framework and issues for Goal and scope definition proposed there. In further operationalising the ISO proposals the work of SETAC Working Groups and relevant proposals by other authors have been taken into due account. Deviations from ISO have been introduced only when there is significant justification for doing so.
We shall now discuss the substance of each of the last three steps distinguished above, thereby following the fixed format: Topic, Developments in the last decade, Prospects, Conclusions and Research recommendations. As explained in Section 1.5, the Procedures step is not discussed separately in the present chapter but in integrated fashion, for all phases, in Section 1.3.

2.2 Goal definition

TOPIC

In the first step of Goal and scope definition the goal of the LCA study is stated and justified, explaining the goal (aim or objective) of the study and specifying the intended use of the results (application), the initiator (and commissioner) of the study, the practitioner, the stakeholders[1] and for whom the study results are intended (target audience). As this step provides the basic starting point for conducting the LCA study, it should make clear the reasons for undertaking the study. This step is important and mandatory for each and every LCA study, not only because the stated application will affect the course of the entire study but also to guarantee clear external communications following completion of the study (Heijungs *et al.*, 1992).

> *Note* that recommendations for appropriate design of an LCA project are provided Part 2a, Chapter 1. These should be duly studied *before* the actual LCA is commissioned and performed.

DEVELOPMENTS IN THE LAST DECADE

ISO 14041, clause 5.2 states the following requirement for the goal of the study: "The goal of an LCA study shall unambiguously state the intended application, the reasons for carrying out the study and the intended audience, i.e., to whom the results of the study are intended to be communicated."

> Heijungs *et al.* (1992)
> The goal definition in Heijungs *et al.* (1992) was considered to be part of the overall goal definition. The overall goal definition referred to that part of the study that established not only the environmental goals of the LCA study but also its economic, financial, product safety, social (e.g. employment) etc. goals. The '92 Guide was concerned solely with *environmental LCA*, as is this guide (*cf.* Chapter 1). Heijungs *et al.* (1992) consequently focused on the environmental reasons for performing an LCA study.

Although not formulated as a requirement under the goal of the study, the ISO standards make a distinction between comparative studies, in particular comparative assertions, and non-comparative studies. Specific (mandatory) requirements are formulated for LCA studies used to make a comparative assertion that is disclosed to the public. In ISO 14040 (1997E) a comparative assertion is defined as "an environmental claim regarding the superiority or equivalence of one product versus a competing product which performs the same function".
On several occasions ISO also mentions the importance of realising the possibilities and limitations of the LCA instrument, both in general and in relation to other environmental assessment tools. Once the type of application has been determined, it is important to establish whether LCA is the most appropriate instrument for answering the specific research question, or whether an alternative tool is

[1] Referred to in ISO 14040 (1997E) as "interested parties".

perhaps more suitable for the purpose or may yield relevant additional information. This choice can be grounded in the possibilities and limitations of LCA in relation to those of other environmental assessment tools (see also appendix B of the present Part; Wegener Sleeswijk *et al.*, 1996; Finnveden, 1998; Finnveden, 2000; Wrisberg *et al.*, in prep.).
It has become increasingly clear that LCA is but one tool amongst many for environmental analysis. In some circumstances other instruments may be more appropriate: Risk Assessment (RA), for example, if siting aspects are crucial to the issue under investigation. Other tools may be used in parallel to provide better insight into the environmental consequences of a given choice: Substance Flow Analysis (SFA), for example, if one specific flow is of prime importance in the product system investigated, as in the case of rechargeable cadmium batteries. The EU-concerted action CHAINET has dealt with this subject extensively (Wrisberg *et al.*, in prep.).

Other authors support the importance of the aforementioned issues, sometimes usefully expanding them into more detailed guidelines. For example, Lindfors *et al.* (1995a) argue that it is not sufficient merely to define the goal in terms of what is to be done, e.g. 'To compare the environmental impacts associated with paints used for wall decoration', but that the underlying reason, e.g. 'To provide information in the setting of criteria for ecolabeling', also needs to be addressed. They suggest including in the LCA report a clear statement of the intended applications and users. For the particular case of paint they give the following example: 'The results of the LCA will be used by the ecolabeling board to identify areas where criteria should be set in order to promote the most environmentally friendly products within this product group'.

Several authors note, furthermore, that it may be necessary to revise the initial objectives and intended decisions during the course of an LCA (e.g. ISO 14041, 1998E; Lindfors *et al.*, 1995a; Wenzel *et al.*, 1997). Performing an LCA is an iterative rather than purely sequential process.

Finally, various references distinguish between different categories of application (e.g. Heijungs *et al.*, 1992; Weidema, 1993; Fleischer *et al.*, 1995; Guinée, 1995; Lindfors *et al.*, 1995a; Braunschweig *et al.*, 1996; UNEP, 1996; ISO 14040, 1997E; Cowell *et al.*, 1997; UNEP, 1999; Wenzel, 1998). The differences between these various categorisation schemes are mainly a matter of taste and not examined as such here. The key question is, rather, whether it is possible to distinguish groups of applications differing in their impact on methodological choices or procedural arrangements in the course of an LCA study. Some authors offer valuable suggestions for a classification of applications as they affect methodological choices (Frischknecht, 1997; Frischknecht, 1998; Clift *et al.*, 1998; Weidema, 1998a; Wenzel, 1998). This subject will be discussed in more detail in Section 2.3 ('Mode of analysis').
This Guide distinguishes six types of decision situation in which LCA results may be applied (De Bruijn & Van Duin, 1998; Van Duin & De Bruijn, 1998):
- global exploration of options;
- company-internal innovation;
- sector-driven innovation;
- strategic planning;
- comparison;
- comparative assertion disclosed to the public.

In the case of comparison of product alternatives, it must be determined at some stage what differences in results are to be deemed significant for concluding that one alternative is environmentally sounder than another. For procedural reasons it seems wise to address this issue as part of the goal definition, at the very outset of the study.

Clause 8 of ISO 14041 (1998E) specifies that the study report on the Goal and Scope definition should meet the following requirements (see textbox):

The results of an LCI study shall be fairly, completely and accurately reported to the intended audience as described by the relevant parts of clause 6 of ISO 14040:1997E. If a third-party report is required, it shall cover all items marked with an asterisk. All additional items should be considered.

a) Goal of the study
   1) reasons for carrying out the study *;
   2) its intended applications *;
   3) the target audiences *.
b) Scope of the study:
   1) modifications together with their justification;
   2) function:
      i) Statement of performance characteristics *;
      ii) Any omission of additional functions in comparisons *;
   3) functional unit:
      i) Consistency with goal and scope *;
      ii) Definition *;
      iii) Result of performance measurement *;
   4) system boundaries:
      i) Inputs and outputs of the system as elementary flows;
      ii) Decision criteria
      iii) Omissions of life cycle stages, processes or data needs *
      iv) Initial description of the unit processes;
      v) Decision about allocation;
   5) data categories:
      i) Decision about data categories;
      ii) Details about individual data categories
      iii) Quantification of energy inputs and outputs *;
      iv) Assumptions about electricity production *;
      v) Combustion heat *;
      vi) Inclusion of fugitive emissions;
   6) criteria for initial inclusion of inputs and outputs:
      i) Description of criteria and assumption *;
      ii) Effect of selection on result *;
      iii) Inclusion of mass, energy and environmental criteria (comparisons *);
   7) data quality requirements.

For Goal and Scope definition, Lindfors *et al.* (1995a) list the following reporting issues:

- The members of the reference panel or review group shall be reported, if relevant.
- A short resumé of the discussions in the reference panel shall be given, with focus on conflicting views,
  or: a report from the reviewer(s) on the critical review, concerned party review, or validation,
  or: a statement that an external validation or review process has not been carried out, including a justification of that decision (e.g. since concerned parties have been involved in the conduct of the study).
- The commissioner of the study shall be stated.
- A presentation of the practitioners, including their background, shall be given (an LCA-oriented C.V.).
- The purpose shall be clearly and unambiguously stated, in terms of the reasons for carrying out the LCA.
- A clear statement on the decisions intended to be based on the findings should be made.
- A statement on the intended users or audience should be made.
- The main user function(s) (e.g. protection and/or colouring for paints) forming the basis for the LCA shall be clearly defined and reported.
- Any deviation from initial plans may be reported.
- Any other limitations or introduced assumptions relevant to the results of the study shall be reported.

In its report, the SETAC-Europe Case studies Working Group (Meier *et al.*, 1997) gives the following (minimum) reporting guidelines for the Goal and Scope phase:

General study information

- As much detail as possible should be included, i.e. authors, affiliations of authors, commissioning body, responsible person at commissioning body, availability, etc.

Goal definition

- The overall objectives of the study should be given in a clear and concise statement with the reasons for carrying out the study and the intended use of the results detailed.
- The methodology employed should be clearly stated and transparent and any differences in methodology from a full LCA should be explained. All assumptions and value judgements should be clearly detailed along with the justification for the assumptions.

These reporting guidelines have also been provided by the Working Group in the form of a practitioners' checklist.

PROSPECTS

It might be possible as well as useful to define categories of LCA applications leading to different choices vis-à-vis methodological procedures and for which different Guidelines would therefore be applicable. This might result in better and more consistent results, and in more time- and cost-effective LCA studies.

CONCLUSIONS

We recommend following the ISO 14041 requirements, supplemented by the suggestions of Lindfors *et al.* (1995a). In this Guide we furthermore distinguish six different types of decision situations:

- global exploration of options;
- company-internal innovation;
- sector-driven innovation;
- strategic planning;
- comparison;
- comparative assertion disclosed to the public.

In the case of comparison of product alternatives, we recommend establishing in the Goal definition step what differences in results are to be deemed significant for concluding that one alternative is environmentally sounder than another.

RESEARCH RECOMMENDATIONS

Short-term research

- Definition of categories of applications, with preferred choices of LCA methodology and associated sets of methodological and procedural Guidelines.
- Standards-setting, possibly by government, regarding the quality and methodology for LCAs for each of these different applications (i.e. a *code of practice* for each application).

2.3 Scope definition

TOPIC

In the scope definition step the main characteristics of an intended LCA study are established, covering such issues as temporal, geographical and technology coverage, the mode of analysis employed and the overall level of sophistication of the study. A so-called goal and scope report may also be drafted for the sake of critical review and comments from interested parties. This report should justify all the main choices with respect to the step Function, functional unit, alternatives and reference flow‘ and the phases of Inventory analysis, Impact assessment and Interpretation.

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)

In Heijungs *et al.* (1992) scope definition did not yet constitute a separate method step. However, the level of sophistication, or depth of the study as it was then called, comprised a separate element of goal definition. A distinction was made between a more detailed and a more streamlined approach. A detailed LCA was considered to be appropriate for important applications such as government approvals or bans. A more simplified approach was considered to be appropriate for certain in-company applications and in LCAs relating to product improvement and design. Simplification was to be achieved by:

- concentrating on the differences between product alternatives;
- excluding certain elements of the LCA;
- limiting the number of processes;
- limiting the number of environmental effects.

It was noted that introducing simplifications might imply decreased reliability, particularly if it is decided to limit the number of processes or environmental effects examined, and that such reduced reliability should be in conformity with the importance of the application. In Heijungs *et al.* simplified methods were not elaborated in any further detail.

The spatial and temporal dimension of the study goal were treated as separate steps in Heijungs *et al.* With regard to spatial representativeness it was noted that "this must be specified unless it is clear from the specification of the functional unit. The spatial representativeness could be global, continental (e.g. European), regional (e.g. EU), national (e.g. the Netherlands) or at company level (e.g. brand *X*). Also, representativeness like 'in temperate climates' may be useful, e.g. in an LCA on insulation materials. The particular spatial representativeness determines which alternative systems can usefully be considered in an LCA study for a specific application (e.g. an LCA study on milk packaging for ecolabeling in the Netherlands may involve different packaging alternatives from the same study for an EU ecolabel)."

With regard to temporal representativeness Heijungs *et al.* states that this should be determined similarly to spatial representativeness. Generally, a rough indication will suffice, for example '1998' or '2010'. Also, representativeness like 'during summer' may be useful, especially for seasonal products. Similarly to spatial representativeness, temporal representativeness determines which alternative systems may usefully be compared in an LCA study for a specific application.

Finally, Heijungs *et al.* already discussed a hot issue in the present LCA debate, which we shall henceforth refer to as the "mode of analysis". A basic assumption made by Heijungs *et al.* (p.12 of Backgrounds '92) is that the *ceteris paribus* principle is applied in product assessments, which means that the choice of the functional unit of the product alternative investigated has no influence on any other

ISO 14041 (1998E), clause 5.3.1 states the following concerning the scope of the study (see textbox):

The scope of the study shall consider all relevant items in accordance with ISO 14040: 1997, 5.1.2.

It should be recognised that an LCA study is an iterative technique, and as data and information are collected, various aspects of the scope may require modification in order to meet the original goal of the study. In some cases, the goal of the study itself may be revised due to unforeseen limitations, constraints or as a result of additional information. Such modifications, together with their justification, should be duly documented.

ISO 14040 (1997E), clause 5.1.2 reads (see textbox):

> In defining the scope of an LCA study, the following items shall be considered and clearly described:
> – the function of the system
> – the functional unit
> – the system to be studied
> – the system boundaries
> – allocation procedures
> – the types of impact and the methodology of Impact assessment and subsequent Interpretation to be used
> – data requirements
> – assumptions
> – limitations
> – the initial data quality requirements
> – the type of critical review, if any, and
> – the type and format of the report required for the study.
>
> The scope should be sufficiently well defined to ensure that the breadth, the depth and the detail of the study are compatible and sufficient to address the stated goal.
> LCA is an iterative technique. Therefore, the scope of the study may need to be modified while the study is being conducted as additional information is collected.

The Scope definition is a relatively new step that has not yet received much attention in other LCA literature. As explained in Section 1.4 and taking into account the elements distinguished as part of Scope definition in ISO 14040 and 14041, we propose distinguishing two main elements of the scope of the study:

– Determining the *main characteristics* of an intended LCA study: temporal, geographical and technology coverage, coverage of economic processes, coverage of environmental interventions and impact categories, mode of analysis and level of sophistication of the study.
– *Reporting* of all the main choices to be made in the Function, functional unit, alternatives and reference flow' step and the Inventory analysis, Impact assessment and Interpretation phases. This yields a so-called Goal and Scope Report, drawn up for critical review and comments from interested parties.

These two elements together cover all the ISO issues mentioned in the above text box. Below, the developments of the past decade are discussed as they relate to the main characteristics: temporal, geographical and technology coverage, mode of analysis and level of sophistication of the study. In Part 2a, guidelines are provided for drafting a Goal and Scope Report in Section 1.6 on reporting.

Temporal coverage

ISO 14041 (1998E), clause 5.3.6 states that temporal coverage refers to:

– The desired age of data (e.g. within the last five years) and the minimum length of time over which data should be collected (e.g. one year).

According to ISO 14041 (1998E) the temporal coverage shall be specified and appropriate data quality requirements defined. The latter point is treated in Section 3.5 of the Inventory analysis.
Temporal coverage has become even more important in recent years, as there is a general feeling today that different modes of analysis should be employed for different temporal contexts (see also 'Mode of analysis', below). It is thus important to specify a base year or base period for the study, on which other choices can be based.

Geographical coverage

ISO 14041 (1998E), clause 5.3.6 states that geographical coverage refers to:

– Geographical area from which data for unit processes should be collected to satisfy the goal of the study (e.g. local, regional, national, continental, global).

According to ISO 14041 (1998E) the geographical coverage shall be specified and appropriate data quality requirements defined. The latter point is treated in Section 3.5 of the Inventory analysis.
In 1996 a 'Groupe des Sages' published a report on the use of LCA for ecolabeling applications in the EU. They stressed the need to explicitly establish the geographical coverage of an LCA study in certain cases, giving the example of regional differences in water hardness influencing detergent requirements. Regional differences might thus be taken into account in determining the amount of detergent needed to

wash a certain amount of laundry optically white. Depending on the precise application of the LCA, this may be a very useful thing to do since it will indicate to consumers how they can minimise the environmental impact of their behaviour, taking into account the regional background (Udo de Haes *et al.* ,1996). The usefulness of including this geographical context depends on the application in question, however. While this may be very useful in a national study, it may not be relevant for a European-average study.

Technology coverage

ISO 14041 (1998E), clause 5.3.6 states that technology coverage refers to:

- Technology mix (e.g. weighted average of the actual process mix, best available technology or worst-operating unit).

According to ISO 14041 (1998E) the technology coverage shall be specified and appropriate data quality requirements defined. The latter point is treated again in Section 3.5 of the Inventory analysis.

The technology coverage taken as the point of departure for data collection in the Inventory analysis should match the geographical and temporal coverage of the study. In a European-average study for the year 2000, for example, data on the European state-of-the-art technology being installed in 2000 should be taken; in a Dutch study for that year, the state-of-the-art in the Netherlands for the year 2000 should be used. Thus, for most structural, change-oriented decisions (see below) the data taken should be for the current state-of-the-art technology in the region or country with which the study is concerned (see also Section 3.6).

If the LCA study concerns a futuristic system for which technologies are available only as prototypes, it is important to determine the criteria for allowing a prototype technology to be taken into account in the LCA study. For example, it may be important to establish that prototypes should have undergone a certain amount of testing before being eligible for the LCA study and that they are economically feasible[1]. With some applications it might also be useful for interested parties to come to arrangements for dealing with implementation of prototype technologies after the LCA study (see Section 1.3 on 'Management of LCA projects: procedures'). As far as is known to the authors of this report, no guidelines have yet been formulated on this issue.

Coverage of economic processes (initial system boundaries)

Life cycle assessment is defined as a compilation and evaluation of the inputs and outputs and the potential environmental impacts of a product system throughout its life cycle (ISO 14040, 1997E). Ideally, the product system should be modeled in such a manner that all the inputs and outputs at its boundary are environmental interventions (ISO 14041, 1998E). However, if it is possible to narrow the scope of the life cycle to those stages or subsystems that really matter in relation to the goal of the study, a significant saving of the time and resources spent on an LCA study can be achieved. Todd *et al.* (1999) and also Christiansen *et al.* (1997) mention that such narrowing down is feasible within the ISO (quantitative) LCA framework by limiting or eliminating upstream and/or downstream stages ("exclude parts of the life cycle system").

A useful first step in any LCA study, whether simplified, detailed or extended, is to consider whether limiting or eliminating certain life cycle stages (without introducing a mere cut-off) is justified in relation to the goal of the LCA study. A 'mere cut-off' here refers to certain life cycle stages or subsystems, e.g. capital goods or inputs contributing less that a certain percentage to a product or process, being excluded from a study because of time and resource constraints. This subject is treated below ('Level of sophistication') and in Section 3.8, but it is not the subject of the present section. Limiting or eliminating life cycle stages may be permissible if the goal of the study allows for a narrower scope with respect to system boundaries without the reliability of the results being reduced. In a study aiming to supply data on a subsystem like aluminium or polyethylene production, for example, it may be justified to narrow the analysis to a cradle-to-gate analysis (i.e. an analysis not going beyond the aluminium or polyethylene production site). Similarly, a gate-to-gate analysis (i.e. an analysis of polyethylene film production from polyethylene granulate) can in some cases be performed, narrowing the boundaries even further. The various limitations of these partial LCA studies should obviously be fully realised by the practitioners and users of the ensuing results. Although these studies may serve as an important and useful source of data for other cradle-to-grave LCAs, in themselves they by no means constitute full LCAs. Any conclusions drawn on the basis of such partial LCAs should therefore be formulated with the greatest caution.

[1] What is meant by "economically feasible" should then also be established, of course.

It may be in order to narrow the boundaries of an LCA for quite different reasons, viz. in the type of LCA here referred to as 'difference analysis'. In this case it may be opted to exclude from further analysis life cycle stages or subsystems that are similar for each of the (product) systems analysed. This holds only for comparative LCAs, where the aim is to focus on and assess the differences between alternative (product) systems. For example, if the filling and distribution processes for two products are similar, these activities might be excluded from the analysis[1]. However, if one of the products requires refrigeration during filling and distribution activities, with higher energy consumption as a result, then these activities should clearly be addressed (Christiansen *et al.*, 1997). An important condition here is thus that the subsystems excluded are precisely comparable with respect to technologies, properties, qualities and quantities of the flow supplied or produced. An example for which exclusion of subsystems is rather more debatable is food packaging. An item frequently excluded from comparative packaging studies is the product packaged. However, due care should be taken in excluding the product, particularly when product waste or actual consumption is influenced by the characteristics of the packaging (Kooijman, 1993; Heijungs & Guinée, 1995; Christiansen *et al.*, 1997).

Coverage of environmental interventions and impact categories

ISO 14040 (1997E), clause 5.1.2.2 states that "the selection of inputs and outputs […] shall be consistent with the goal of the study". ISO 14041 (1998E), clause 5.3.4 states that "individual data categories [including environmental interventions - authors' addition] should be further detailed to satisfy the goal of the study". In addition ISO 14042 (2000E), clause 5.3 states that "the selection of impact categories shall reflect a comprehensive set of environmental issues related to the product system being studied, taking the goal and scope into consideration". Although LCA, as defined in this Guide, in principle aims for a broad coverage of environmental interventions and impact categories (see Chapter 1), these ISO quotations leave some scope for restriction.

Todd *et al.* (1999) as well as Christiansen *et al.* (1997) mention two main options for narrowing the coverage of environmental interventions and impacts:

- focusing on specific environmental impacts or issues (CO_2, for example);
- establishing criteria to be used as 'showstoppers' or 'knockouts', e.g. criteria which can result in an immediate decision when encountered during the study (a black list chemical, for example).

Focusing on certain inputs or outputs, in an energy analysis for example, or on certain environmental interventions and/or specific impact categories may be very practical and defensible in the context of a specific goal, but the drawback is that important environmental factors may thus be excluded. The most comprehensive approach is, of course, to include all environmental interventions and impact categories listed by the SETAC-Europe Working Group on Impact assessment (Udo de Haes *et al.*, 1999). Those environmental interventions and impacts that cannot be quantified for lack of appropriate methodologies and/or data should then be handled in a qualitative manner (and flagged where relevant), developing additional methods and/or calculating additional factors where possible. For a number of categories practical methods are as yet lacking, however. Thus, the best available practical option is to include all impact categories for which practical methods are available and all associated environmental interventions for which factors are available. Environmental interventions encountered in a study for which factors are lacking but which are deemed important in relation to the impact category considered should be handled qualitatively. An extended option here is to calculate new factors for such environmental interventions.

Mode of analysis

In general, the term 'mode of analysis' might be taken to refer to the specific manner in which methods are employed to answer a particular question. For instance, one might address a certain problem scientifically, qualitatively, dynamically or whatever the case may be. In the context of LCA, however, we employ the term to refer to the two main choices described Section 1.1[2]

- change-oriented versus descriptive analysis;
- occasional versus structural versus strategic choices.

[1] Another example is waste management. If a waste stream for which two management techniques are being compared is exactly the same for both techniques, the upstream waste management processes may be excluded from further analysis.

[2] In that section, we also listed a number of synonyms or related terms that can be found in literature, like prospective LCA.

In Section 1.1, we chose to focus the discussion and the resulting guidelines on change-oriented analysis in support of structural decisions. Relevant examples are a choice between packaging materials (for a company), a choice between means of transport (for a frequent traveller) and so on. It was stated that the other modes of analysis, for occasional choices or by means of a descriptive analysis, for example, may be relevant in certain decision contexts. The emphasis on change-oriented analysis and structural decisions was motivated by the need to restrict the topic, on the one hand, and by the broad scope of our interest, on the other. In certain decision contexts this mode of analysis may be inappropriate. In others, it may provide a coarse approximation to the question the LCA is designed to answer. It is difficult to specify exactly what limitations this sets on the scope and validity of the theory developed in the present documents and the derived guidelines. In most cases it will suffice for practitioners to provide a qualitative justification of how the study satisfies the terms of change-oriented, structural LCA elaborated in this Guide.

With respect to the Scope definition phase, one practical consequence is that the assumptions behind such change-oriented analysis and structural decisions must be justified as being reasonable. It also means that a bad match vis-à-vis these assumptions may lead to results that are inaccurate and of only limited relevance. For instance, a question like 'Shall I take the car or train for my business trip tomorrow' is not related to a structural choice and the method described in this Guide may give an erroneous answer when applied to such a question. In the Scope definition it is therefore important to indicate relevant mismatches and effects that may potentially lead to reduced accuracy and restricted usefulness. This obviously cannot take the form of quantitative analysis complete with in-depth error analyses. It will be no more than a qualitative description of possible restrictions with uncertain consequences. However, it may also lead to recommendations to address particular gaps in knowledge. In certain cases, for instance, it might be argued that the topic is more amenable to short-term optimisation using the tools of operations research (e.g. linear programming, *cf.* Clift *et al.*, 1998).

Level of sophistication

The fact that LCA is commonly perceived as being extremely complex, time-consuming and expensive may discourage potential users (Christiansen *et al.*, 1997). However, it is generally acknowledged that LCA does not refer to one specific method but rather to a framework for systematic and comprehensive environmental assessment of (product) systems. The SETAC-Europe LCA Screening and Streamlining Working Group (Christiansen *et al.*, 1997) and the SETAC North America Streamlined LCA Workgroup (Todd *et al.*, 1999) have suggested that different levels of sophistication of LCA may be required for different decision situations. In line with these suggestions, and to promote broader application of LCA, guidelines might be developed for different levels of LCA sophistication.

Christiansen *et al.* (1997) suggest distinguishing between simplified LCA and detailed LCA, providing the following definitions:

Simplified LCA: an LCA of reduced complexity produced by the procedure of simplification.

Simplification: a procedure to reduce the complexity of an LCA and so reduce the cost, time and effort required to run it. This may entail exclusion of certain life cycle stages, system inputs or outputs, or impact categories, or the use of generic data modules rather than data specific to the system under study.

Simplification was considered to consist of three steps:

- Screening: identification of elements of the LCA that can be omitted or for which generic data can be used without significantly affecting the accuracy of the final result[1].
- Simplifying: application of the simplifying options identified in the screening step to produce a simplified LCA.
- Assessing reliability: ensuring that the results are reliable enough to justify the conclusions drawn.

Streamlined LCA was considered to be synonymous with simplified LCA and streamlining was considered synonymous with simplification, which seems in line with the definitions of the North American Working Group (Todd *et al.*, 1999)[1]:

[1] Screening was defined by Christiansen *et al.* (1997) as a procedure that identifies some particular characteristic or key issue associated with the LCA that will normally be the subject of further, more intensive study. Such key issues might be:

- the principal environmental inputs/outputs or environmental impacts associated with the particular life cycle;
- the stages of the life cycle giving rise to the most significant environmental inputs and outputs;
- major gaps in the available data.

- Using qualitative as well as quantitative data, for example constructing materials-flow diagrams for the processes and materials studied but not always quantifying flows.
- Using surrogate data ('use of secondary data sources'), for example using readily available data on a similar process when data on the particular process cannot be obtained.
- Limiting the process flows studied to those exceeding a certain threshold value.

The first option for simplification is applicable to all LCA studies and has already been discussed. The remaining options relate basically to the problem of the inventory data. The use of surrogate data or, better, estimates, when process-specific data are lacking has recently been extended from use of public data reports and databases to use of data based on input-output analysis: IOA (Lave *et al.*, 1995; Hendrickson *et al.*, 1998). Since this possibility has been operationalised, there are in fact no longer any grounds for excluding any flows from an LCA study (for an extensive discussion, see Section 3.8). Flows for which specific data are lacking can now be estimated using IOA models. This implies that the main simplification with respect to inventory data is to reduce the number of processes for which specific (primary) data are to be collected. These processes will henceforth be referred to as 'foreground processes'. Processes for which no specific (primary) data are collected but for which secondary data from databases, public references or IOA-based estimates are used will be referred to as 'background processes'[2]. The more foreground processes are included in a specific LCA study, the more 'detailed' the LCA will be.

Impact assessment based on a tailor-made factor approach is so straightforward, except for factors and methods that are lacking, that the duration and cost of the LCA study will not be significantly affected.

There are two other issues which are not mentioned by Christiansen *et al.* (1997) and Todd *et al.* (1999) but which are likely to co-determine the amount of time and resources spent on an LCA study:
- procedures, e.g. a critical review;
- reporting, e.g. the frequency of interim reporting and the level of detail of these reports.

In this Guide, from this point onwards individual sets of guidelines will be developed for two levels of sophistication, acknowledging that more than two levels might, of course, be distinguished here[3].
- Detailed LCA: an LCA based on practical guidelines complying with ISO standards, where time and resource constraints do not play a dominant role; detailed LCAs are based on the assumptions and simplifications described in Section 1.2.2.3.
- Simplified LCA: an LCA based on practical guidelines not fully complying with ISO standards, e.g. standards on data collection, data quality requirements, data validation and allocation, for reasons associated with time and resource constraints; the simplified level has been introduced to provide a basis for performing faster and cheaper LCAs than detailed LCAs.

For detailed LCA there are also options for extension.
- Options for extension: options for extending a detailed LCA to include, for example, new but not yet fully operational technologies (i.e. sensitivity analyses on these), elaboration of new impact categories, development of new characterisation and normalisation methods and/or calculation of associated factors, partial uncertainty analyses and so on. Note that applying these options for extension implies a deviation from the assumptions and simplifications described in Section 1.2.2.3 and from the practical points of departure of each LCA phase, as described in Part 2a.

There is no specific set of guidelines for extended LCA, as there is for simplified and detailed LCA, but certain additional guidelines or suggestions are here provided for extending several individual steps of a detailed LCA[1].

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[1] Todd *et al.* (1999) define screening LCA as an application of LCA used primarily to determine whether additional study is needed and the required focus of that study. They define streamlined LCA as the identification of elements of an LCA that can be omitted or for which surrogate or generic data can be used without significantly affecting the accuracy of the results.

[2] These pragmatic definitions deviate from other definitions such as those in the report of the SETAC Working Group on Inventory enhancement (Clift *et al.*, 1998).

[3] Simplified LCA and detailed LCA with extensions are considered here to be reasonable extremes of the possible range of quantitative LCAs. Of course, further simplification is possible, e.g. a qualitative scanning LCA (used in some ecolabeling programmes) as well as further sophistication, e.g. an LCA study in combination with a Risk Assessment study. These levels of LCA levels are not addressed here further, however.

Note that the definitions used in this report differ from those of Christiansen *et al.* (1997) and Todd *et al.* (1999). This is because streamlined and simplified LCA as defined by these respective authors are not so much methods as procedures or applications of LCA. In this Guide we have aimed to develop different sets of methodological guidelines for these different types of LCA. The definitions have therefore been adapted somewhat.
For the six decision situations distinguished in Section 2.2, we make the following recommendations with respect to the preferred level of sophistication:

Table 2.3.1: Recommended level of sophistication for six different decision situations.

	simplified	detailed	extended
Global exploration of options	X		
Company-internal innovation	X		
Sector-wide innovation	X	X	
Strategic planning	X	X	
Test/comparison		X	
Comparative assertion disclosed to the public		X	X

As well as steering the procedural part of a study (see Section 1.3 of this Part of the Guide and Part 1 and 2a), because it will largely determine the appropriate level of sophistication, the decision situation may also be of decisive influence on the methodological choices made within an LCA study.
Clearly, Scope definition is a crucial step that has major technical and procedural consequences. Decisions taken during this step will have a huge bearing on the significance of the LCA results and it is therefore important to reflect accordingly on the issues for Interpretation.

Whether the question to be answered by the LCA is related to Occasional, Structural or Strategic Choices is of major importance for the choices of models and data later on in the study (see Chapter 1 and Section 2.3 of this chapter). The model used and the data selected should reflect the strategic scope of the decision the LCA is intended to support. As starting points for this Guide, a number of simplifications have been made with respect to time (in model relations and in the specification of inventory results), geographical aspects (e.g. location of processes and impacts) and extent of the mechanisms included (e.g. fixed input/output relations, technical production function, market and social mechanisms). If these simplifications would put severe constraints on the validity of the study results, they should be reconsidered and if possible adjusted. Thus, it should for example be validated whether spatial information can justifiably be excluded from the study.
It should be duly validated whether temporal, geographical and technology coverage and coverage of economic processes, environmental interventions and impact categories are consistent with and sufficiently comprehensive in relation to the goal(s) defined. The main check to be made with respect to coverage of economic processes is whether phases or subsystems can justifiably be omitted in relation to the overall goal of the study. Furthermore, it should be checked whether the impact categories and environmental interventions selected are the relevant ones for the (product) systems studied and whether they are in line with the goal of the study.
For six decision situations, we provide recommendations above on the most appropriate level of LCA sophistication (simplified, detailed, detailed and extensions). These decision situations are defined in broad terms and which of them matches the specific goal of the intended LCA study most closely should be duly examined. If it is opted to deviate from the recommended level of sophistication, sound reasons should be given for doing so. For example, a simplified LCA approach should not be chosen in an LCA study geared to a comparative assertion unless it is explicitly defined as a preliminary study to a detailed one, with the latter designed to support the comparative assertion in question.

For Goal and Scope definition Lindfors *et al.* (1995a) list the following reporting issues:
- Co-functions excluded from the analysis by use of partitioning (allocation) should be reported.
- Final system boundaries shall be justified in relation to the goal of the study.

[1] According to Christiansen *et al.* (1997), it is not possible to formulate specific guidelines for simplified LCA that are applicable to all types of product systems. Although this may be true, we deem it feasible to develop generic guidelines, which need to be elaborated further in each LCA study, on how to simplify certain methodological elements.

- Temporal boundaries for impacts should be given and justified.
- The final choice of impact categories, level of assessment and omitted categories and flows shall be reported according to the classification list given in this guideline [in this Guide, see Section 4.2].
- Maximum base-line data quality requirements should be justified.
- Any deviation from initial plans may be reported.
- Any other limitations or assumptions of relevance to the results of the study shall be reported.

Meier *et al.*, 1997 give the following reporting guidelines for the scope phase:
- The boundaries, methodology, data categories and assumptions should be clearly stated and understandable. The scope of the study should be defined in sufficient detail to enable the study to address the stated objectives including stages of the life cycle, system boundaries, methodology, data requirements and assumptions.
- Data quality goals and any variability of data considered in the study should be clearly established and detailed.
- The methodology used should be clearly and transparently stated and any differences in methodology from a full LCA should be explained. All assumptions and value judgements should be clearly detailed along with the justification for the assumptions.

These authors also provide these reporting guidelines in the form of a practitioner's checklist.

PROSPECTS

The discussion on descriptive and change-oriented (retrospective – prospective) analysis is still continuing and it is expected that this will give rise to new ideas in the nearer and more distant future.
No specific developments are foreseen in the other areas.

CONCLUSIONS

Two elements can be distinguished with respect to the Scope of the study:
1. the main characteristics of the study, including temporal, geographical and technology coverage, coverage of economic processes, coverage of environmental interventions and impact categories, mode of analysis and level of sophistication of the study; and
2. reporting, resulting in a Goal and Scope Report.

The step Scope definition deals only with the main characteristics of the intended LCA study. Issues to be covered by a Goal and Scope Report are discussed elsewhere.

With respect to these main characteristics of an LCA the following recommendations are made:

temporal coverage
specify base year or period according to ISO 14041

geographical coverage
specify geographical area according to ISO 14041

technology coverage
specify technology coverage according to ISO 14041; take current state-of-the-art technology as the baseline

coverage of economic processes
include all economic processes; narrow the number of economic processes included in cradle-to-gate, gate-to-gate or a difference analysis

coverage of environmental interventions and impact categories
include all impact categories for which practical methods are available and all environmental interventions for which characterisation factors are available, unless the scope is explicitly narrowed to a few environmental interventions and/or impact categories; include remaining impact categories and interventions qualitatively as far as possible ('flags')

mode of analysis
change-oriented analysis for structural decisions

level of sophistication
distinction between detailed, simplified and options for extensions

RESEARCH RECOMMENDATIONS
No research recommended.

2.4 Function, functional unit, alternatives and reference flows

TOPIC
In this step the function, functional unit, alternatives and reference flows are defined. The functional unit describes the primary function(s) fulfilled by a (product) system, and indicates how much of this function is to be considered in the intended LCA study. It will be used as a basis for selecting one or more alternative (product) systems that might provide these function(s). The functional unit enables different systems to be treated as functionally equivalent and allows reference flows to be determined for each of them.

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)
In Heijungs *et al.* (1992) this topic was treated under the heading "Defining the subject of the study" (section 1.3 of 1992 Guide). It was stated there that quantitative terms can be included in the process tree once a functional unit has been selected, implying that quantification of the reference flow was considered to be part of the Inventory analysis. The selection of the (product) systems to be compared and considered equivalent on the basis of the functional unit defined, was also an explicit element of the step "Defining the subject of the study" in Heijungs *et al.* (1992).

ISO 14041 (1998E), clause 5.3.2 states the following with regard to the topic of function and functional unit (see textbox):

In defining the scope of an LCA study, a clear statement on the specification of the functions (performance characteristics) of the product shall be made.
The functional unit defines the quantification of these identified functions. The functional unit shall be consistent with the goal and scope of the study.
One of the primary purposes of a functional unit is to provide a reference to which the input and output data can be normalised (in a mathematical sense). Therefore the functional unit shall be clearly defined and measurable.
Having defined the functional unit, the amount of product, which is necessary to fulfil the function, shall be quantified. The result of this quantification is the reference flow.
The reference flow is then used to calculate the input and outputs of the system. Comparisons between systems shall be done on the basis of the same function, quantified by the same functional unit in the form of their reference flows.
If additional functions of any of the systems are not taken into account in the comparison of functional units, then these omissions shall be documented. For example, systems A and B perform functions *x* and *y* which are represented by the selected functional unit, but system A also performs function *z* which is not represented in the functional unit. It shall then be documented that function *z* is excluded from this functional unit. As an alternative, systems associated with the delivery of function *z* may be added to the boundary of system B to make the systems more comparable. In these cases, the processes selected shall be documented and justified.

Source: ISO 14041, 1998E.

Based on these ISO 14041 requirements, the following steps are distinguished in ISO/TR 14049 (1998) for defining a functional unit and determining the reference flows:
- identification of functions;
- selection of one or more functions as the relevant one(s);

– defining the functional unit;
– determining the reference flow.

Below these four steps are described, based largely on ISO/TR 14049 (1998).

Identification of functions
In this step the purpose served by the product system, i.e. its function or functions, is identified. The starting point for this procedure may be a specific product to be studied (e.g. wall paint) or it may be the final need or goal, which may sometimes be fulfilled by several distinct products (e.g. wall decoration, which may be fulfilled by both paint and wallpaper or a combination of these). The functions are related to specific product (e.g. packaging) or process properties (e.g. transport), each of which may:
- fulfil specific needs and thereby have a use value, which typically creates economic value to the supplier of the product,
- affect the functioning of other economic systems (e.g. wallpaper may have a - small - insulation effect, thus affecting the heat requirements of the building).

Selection of function(s)
In this step the relevant functions are selected, on the basis of which a functional unit will be defined and equivalent (product) systems selected in the case of a comparative study. The functions identified in the first step need not all be relevant for a particular LCA study. Thus, out of all the possible functions those that are relevant must be identified. For a solid interior wall, for example, surface protection may be unnecessary, while colouring is a relevant function of paint. See the textbox for more examples.
The result of this step will determine which system alternatives will have to be taken into account and which not in the specific LCA study. In a comparative study, particular care will therefore have to be taken to ensure that any additional functions of each alternative are duly identified and described, and that all relevant functions are taken into account.

Beverage packaging
Beverage bottles have a packaging and an image function. The latter might also be relevant to take into account in some cases. Leaving out the image function of the packaging may lead to comparison of packagings that are technically similar (i.e. containing the same volume of beverage) but which the producer or customer will not accept as comparable.

Hand drying
Hand drying systems have a drying and a hygienic function. Both may be relevant to take into account. Leaving out the hygienic function may be regarded as unacceptable, e.g. in the food industry, where the bacteria-removing ability of paper towels may be regarded as such an advantage that a comparison to electrical hand drying systems may not even be considered.

Source: ISO/TR 14049 (1998).

If a system fulfils just one specific function, the selection of function(s) step will be fairly straightforward. However, systems often fulfil more than one function. For instance, the primary function of shampoos is to clean hair but other important functions may include aesthetic (hair brightness) or hygienic (dandruff) concerns, or communicative functions (just smell how I washed my hair for you). Udo de Haes *et al.* (1996) distinguished two kinds of multifunctionality:
– the product system is defined by its primary function (e.g. washing), and all other functions of the product are facultative (e.g. anti-dandruff function);
– the product system is intrinsically multifunctional, with even the simplest 'design' not being able to be reduced to one function (e.g. a food product, see below).
Basically, they proposed two alternative solutions to this multifunctionality problem[1]:
a) **monofunctional approach:** 'only the primary function of the system is considered, with analysis restricted to (one or more) products fulfilling this function, regardless of any other functions they might fulfil;

[1] Originally proposed for application of LCA in the EU's ecolabeling programme, these two approaches have been adapted slightly by the authors of this Guide for use in general LCA applications.

b) **multifunctional approach**: the primary as well as other functions of the system analysed are considered (e.g. normal shampoos, anti-dandruff shampoos etc.).

In the mono-functional approach, more or less recommended as a baseline by Udo de Haes *et al.* (1996), only the predominant function of the product system is taken into account. In the multifunctional approach one can define the function as precisely as one wishes. However, the more strictly the functional unit is described, the fewer alternatives will be left to compare. The functional unit 'watching TV for 1 hour' may be specified to include more and more functions, as in 'watching colour TV for 1 hour', 'watching large-screen colour TV for 1 hour', 'watching large-screen colour TV with remote control for 1 hour', etc., until there are no product alternatives left to compare. There will generally be various options for drafting an accurate definition of the functional unit which at the same time covers a range of slightly different alternatives and no hard and fast rules can therefore be given on this point.

ISO/TR 14049 (1998) states that "some functions may be so intimately linked that separation is not possible, e.g., the heat generation of a light bulb cannot be detached from its primary function lighting. In other situations, separations of two linked functions may be technically possible, but due to other aspects, the two separate functions may still not be regarded as comparable to the joint functions. An example of this is the combined freezer-refrigerator, which may or may not be compared to a freezer and separate refrigerator, depending on the acceptability of this choice to the consumer (the latter option will typically take up more space than a combined option with the same internal volumes)".

Lindfors *et al.* (1995a) state that the multifunctionality problem should be addressed by either "using a multifunctional basis [for comparison] or by allocation, i.e. partitioning of resources, emissions etc. between the primary function and co-functions not included in the analysis. This shifts the problem to the Inventory analysis, i.e. the allocation."

The conclusion here is that the multifunctionality problem can be solved in several ways:

1. Take into account the primary function only and neglect *all* other functions;
2. Take into account the primary function and all, or a selected number of, additional functions;
3. Allocate between the primary function and the additional functions not included in the analysis.

There is no further rationale for preferring one or other of these options; the appropriate choice simply depends on the goal of the study. Allocation (option 3) can be applied if the functions can be physically separated, as in the case of, say, combined heat and power (co-generation) plant. If separation is not feasible or appropriate, the additional function(s) can be neglected (option 1) or added to the primary function (option 2). If the goal is to compare, say, the cleaning function of a shampoo and nothing else, option 1 should be chosen and all other functions neglected. If the goal is to compare both the cleaning and the anti-dandruff function, option 2 should be adopted and the additional (anti-dandruff) function included in the functional unit. In practice, a product system may fulfil numerous functions at the same time and these examples and options will therefore have to be juggled in such a way as to yield two (or more) primary functions, again permitting one either to allocate for the other functions (option 3) or neglect them (option 1). Whatever the case, it will be clear that the eventual choice is highly dependent on the goal of the study and that the choice made should be duly justified and reported.

The procedural aspect of the 'selection of function(s)' step may be very important. A number of stakeholders may have to be explicitly involved in this step of the study, for otherwise the final results may encounter acceptance problems. For a further discussion of these procedural aspects, see Section 1.3.

Definition of functional unit

Subsequently, the relevant functions are quantified in the functional unit. There are two key issues here: unit and quantity. Unit definition is of course related to the function or goal of the study. Defining the functional unit in terms of mass, mass per surface area or mass per surface area for *X* years will give different units: respectively kg, kg/m^2 and kg/m^2.yr. Any combination of units is in fact possible, depending on the function defined. See the textbox for examples from ISO/TR 14049 (1998) and Wegener Sleeswijk *et al.* (1996).

Wall colouring (ISO/TR 14049, 1998)
For wall colouring, the functional unit will typically have to specify the area to be covered (e.g. 20m^2), the type of wall (especially regarding its absorption and binding properties), the ability of the paint to hide the underlying surface (e.g. 98% opacity), and its useful life (e.g. 5 years).
In the case of multifunctional units, the different quantities are sometimes linked, e.g. a wall covering insulation material may be available with a pre-coloured surface, which makes colouring unnecessary, thus supplying both insulation and colouring. The functional unit might then be:
"20 m^2 wall covering with a thermal resistance of 2 m^2K/W, with a coloured surface of 98% opacity, not requiring any other colouring for 5 years".

Wheat (Wegener Sleeswijk et al., 1996)
An interesting discussion of units in relation to the goal of the study can be found in a methodology report on agricultural LCAs (Wegener Sleeswijk *et al.*, 1996). If two wheat cultivation methods are compared to assess which is the environmentally sounder, the functional unit might be expressed as kg of harvested wheat. However, if the aim of the comparison is to determine the most environmentally sound form of agriculture for a certain geographical area, a better functional unit might be ha.yr wheat cultivation. In the latter case the goal is not to achieve the most efficient form of ‚production but the environmentally soundest use of agricultural soil per year in a given area. These two different goals result in different units. It is another question entirely, of course, whether LCA is at all useful for questions relating to spatial planning (see section 2.2 of this Part and see Part 1).

Food LCAs (Wegener Sleeswijk et al., 1996)
Wegener Sleeswijk *et al.* (1996) give a practical suggestion on how to deal with the multifunctionality problem in defining a functional unit in food LCAs. They state that foodstuffs often fulfil more than one function. They are frequently a source of nutrition and a source of enjoyment at one and the same time. In establishing the functional unit of a product with more than one function, *real substitution* should form the basis for defining an “abstract multifunction”: one functional unit of one product alternative should constitute a real substitute for one functional unit of another product alternative. In a comparison of, say, Wiener schnitzel and steak, it is important to find out how much Wiener schnitzel a consumer would eat in practice as a substitute for a given quantity of steak. If, for example, it is found in practice that an average portion of steak weighs 100 grams and an average portion of Wiener schnitzel 150 grams, the obvious approach would be to compare a portion of steak (100 grams) with a portion of Wiener schnitzel (150 grams). If in practice meat purchase is based above all on mass, however, a comparison of 100 grams of steak with 100 grams of Wiener schnitzel is the more appropriate choice.

The quantity refers to the amount of function or service analysed. Basically, any arbitrary quantity can be taken, since its meaning is only relative in a comparative LCA. For reasons of Interpretation, it may be attractive to make its meaning absolute, e.g. annual (or annual individual) demand for a certain function in the Netherlands[1]. It will be clear that in this case the function will first have to be defined in temporal terms.

Lindfors *et al.* (1995a) and Wenzel *et al.* (1997) state that the durability or the life span of the product should be taken into consideration in relation to the quantity of function/service analysed. In other words, the duration of the function provided by the product system should be taken into account. They give the example of the life span of paint in relation to the time it should protect a specific area, which is comparable to the example of wall colouring in the textbox.
If, in the selection of function(s), more than one function has been selected as being relevant for the comparison in question, a multifunctional unit must be defined. The quantities of the different functions are sometimes linked, e.g. a wall covering insulation material may be available with a pre-coloured surface, which makes colouring unnecessary, thus supplying both insulation and colouring. The functional unit might then be: ‘20 m^2 wall covering with a thermal resistance of 2 m^2K/W, with a coloured surface of 98% opacity, not requiring any other colouring for 5 years’ (ISO/TR 14049, 1998). As noted in ISO/TR 14049 (1998), the equivalence of two products is in many cases determined by user acceptance. This acceptance, and thus whether two products are to be deemed comparable or not, may be influenced by the price of the alternatives and by the additional information given along with the

[1] It should be noted however, that this absolute meaning is generally not valid for all preceding processes.

products, e.g. information on their environmental performance. Thus, for the purposes of product development or strategic management, it may be reasonable to compare two products not intuitively regarded as equivalent but assumed to be so under specific conditions of price and information. User satisfaction may also be determined by a complex of factors, as in the case of food. As this complex of factors is often hard to disentangle, pragmatic solutions are required, as suggested by Wegener Sleeswijk *et al.* (1996), for example; see textbox.

Defining the reference flow

Having defined a functional unit, the next task is to select alternative (product) systems that can provide this functional unit and quantify the performance of these systems using so-called reference flows. In comparative studies both tasks are relevant, while in non-comparative studies only the reference flow of the single (product) system studied need be established.

ISO/TR 14049 (1998) does not explicitly discuss the selection of alternative product systems for analysis in a comparative LCA study. This is often a debating point in such studies, however, and the same frequently also holds for specification of certain key parameters of the reference flow or final product (e.g. trip rate, life span, mass and volume material specifications, etc.).

Lindfors *et al.* (1995a) make an important addition to the ISO standards: in their view it should be mandatory to report not only the system alternatives studied, but also those not studied. As they say: "There might be other alternatives, which are not included in the study. To gain credibility it is important that relevant alternatives that are not studied are commented upon in the report. It is not necessary to list all product or service alternatives not covered by the study, but a minimum requirement is a comment that there are other alternatives and an explanation why these have not been considered."
Furthermore, they argue that "the choice of the reference alternative in comparative studies is a critical issue. An improper choice is obviously likely to give misleading results. Comparison of one product, based on up-to-date information, with a reference product, based on 5–10 years old literature data is a classic example." Comparison of a new design with a reference product based on earlier technologies is similarly improper. In such cases it would be more appropriate to compare the new design to a 'redesigned' reference product, or at least to a reference product updated to up-to-date technology. It will be clear that the choice of the reference product will always affect the result of comparative studies. As Lindfors *et al.* (1995a) state, "it is important that this choice is made in such a way, that it is as representative as possible and that particularly the data qualities of compared alternatives are reasonably consistent."

System performance is quantified by means of a so-called reference flow, i.e. "a measure of the needed outputs from processes in a given product system required to fulfill the function expressed by the functional unit" (ISO 14041, 1998). The reference flow is the connecting flow between the physical output of a system and the amount of function delivered by that system as quantified in the functional unit. It is the flow upon which the whole LCA is based, for example the amount of detergent required (= reference flow) for washing a certain amount of clothes optically white (= functional unit)[1].
Lindfors *et al.* (1995a) argue that a "performance quality standard (**if any**), e.g. a certain level of corrosion protection provided by an industrial paint system" should be taken into account in quantifying the reference flow. This is supported by the work of Udo de Haes *et al.* (1996). In line with the proposal of Udo de Haes *et al.*, three types of system performance can be distinguished:

a) **Standard performance** refers to a known standard (national, e.g. DIN, or international, e.g. CEN), describing a standardised test applicable to all equivalent products: for example, the amount of detergent required to wash to a certain degree of cleanness a certain quantity of clothes with a certain degree of soiledness at temperature *X*, at *Y* degrees of hardness, etc.
b) **Recommended performance** refers to manufacturers' recommendations on product operating mode: for example, the detergent dosage indicated on the packaging.
c) **Actual performance** refers to the actual performance which will often depend on the consumer behaviour, e.g. the average detergent dosage used by the consumer based on consumer studies.

[1] Note that the functional unit and reference flow are different quantities. On the basis of one functional unit, different reference flows will usually be quantified for each (product) system analysed. Only in exceptional cases will reference flows and functional unit be the same, but this will then generally limit the number of products that can be compared.

This actual performance can be taken as an average or as a range, which may a significant factor in the final LCA results. The latter might be used to provide environmental advice to consumers, etc.

Several examples of reference flows are described In the textbox.

Paint

Reference flows are typically expressed as the number of litres required for covering the surface area as defined by the functional unit. For example, in a standardised test, paint A may be determined to cover 8.7 m^2 per litre, thus requiring 2.3 litres to cover the 20 m^2 of the functional unit, provided the conditions in the standardised test are similar to those required by the functional unit (with regard to surface type and opacity).

Hand-drying

In a comparison of paper towels versus an electrical hand drier, it may be irrelevant to use a standardised test based on the technical properties of the paper such as mass, absorption power and tensile strength, if the actual weight of paper used depends on the dispenser design. A more appropriate measure would then be data collected by weighing the paper stock at the start and end of a suitable period in which the number of hands dried are determined by electronic surveillance of actual wash basins located in relevant institutions. Similarly, technical specifications of an electrical hand-drier, such as the volume of air and its temperature, may be irrelevant as a basis for calculating the reference function, if the actual running time of the device is determined by other factors, e.g. a built-in timer. Then, all that is needed is the running time and the electrical capacity of the equipment.

Refrigerators

For long-lived products, such as refrigerators with lifetimes of 10 or 20 years, technology development may be a factor that cannot be disregarded. One refrigerator with a lifetime of 20 years cannot simply be compared to two successive, present-day refrigerators with a lifetime of 10 years. The refrigerators available 10 years from now are certain to be more energy-efficient than current models; the energy efficiency of the second refrigerator in the 10+10 years option must be established by projecting trends, while that of the 20 years option is fixed.

A comparison of refrigerators may be based on their internal and/or external volume. Although the primary function is obviously related to the internal volume, the external volume may a determining factor if the refrigerator is to be fitted in an existing kitchen. If an identical external volume is demanded, the internal volume may differ because of differences in insulation thickness. This can only be adjusted for by assuming differences in user behaviour (e.g. more frequent shopping trips, storage of certain items outside the refrigerator, adding another, smaller refrigerator elsewhere in the house). Each of these changes in behaviour will involve changes in different processes, which will then have to be included in the study. If, on the other hand, an identical internal volume is demanded, a change in insulation thickness may require adjustments in the physical surroundings of the refrigerator (the other kitchen furniture). If both the internal and external volume must be equal, there is obviously no adjustment feasible to accommodate the change in insulation thickness. As this demonstrates, the choice of required functions also determines the possible alternatives to be included in the study.

Beverage packaging

100,000 half-litre one-way bottles may technically fulfil the same function of protecting 50,000 litres of beverage as 125,000 0.4-litre returnable bottles with a reuse rate of 90%. For the consumer, however, the difference in volume may be indistinguishable. If the consumer takes ‘a bottle to be a bottle’, total consumption of the beverage will decrease when the returnable bottles are introduced. In this case, the packaging cannot be studied independent of its contents. In this example there should also be iteration back to the “selection of relevant function(s)” step, or, alternatively, the goal of the study should be redefined, allowing for a comparison of beverage plus packaging taking into account the changes in consumption.

Source: ISO/TR 14049 (1998). For even more examples see also ISO/TR 14049 (1998).

In the case of a multifunctional unit, the reference flows must be quantified in such a way that they fulfil all the functions included in the functional unit. This may imply that, although one of the functions can already be fulfilled using smaller amounts of the reference flow, the amount of reference flow has to be further increased to fulfil the other functions. If this is not acceptable, to the commissioning or interested parties, for example, the multifunctional problem should be solved by allocation (including system expansion) rather than by including the extra functions in the functional unit.

Although the examples and phrasing of ISO/TR 14049 certainly provide a degree of insight, they still do not pin down precisely what a reference flow is or how it relates to a functional unit. Let us therefore expand and focus the discussion a little more. Consider a comparison of two systems for lighting a room with the same amount of light: an incandescent lamp and a fluorescent lamp. Suppose we have defined the functional unit as lighting a standard room for 1 year with a certain flux of light. The flow diagrams of the two product alternatives then contain the two use processes as follows (see Figure 2.4.1).

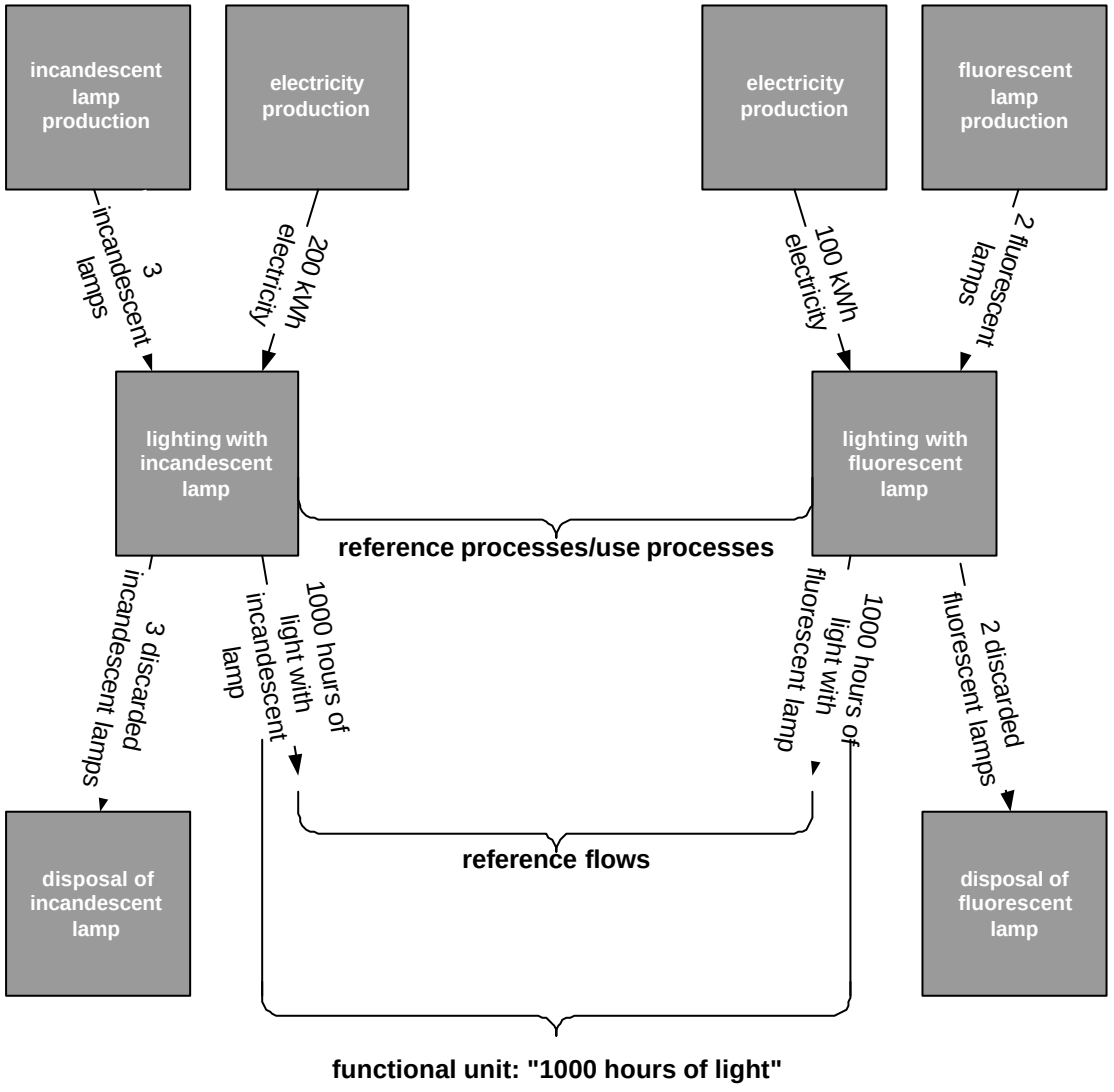


Figure 2.4.1: Flow diagrams, functional unit, reference/use processes and reference flows for two alternative (hypothetical) lighting systems.

Following the ISO examples, the reference flows would be 3 incandescent lamps for the first system and 2 fluorescent lamps for the second. However, lamps are only one of the input flows to the respective use processes, electricity being the other. It might therefore equally well be argued that the reference flow for the first system is 200 kWh of electricity, for the second 100 kWh. Or one could say there is a set of reference flows for each system. Another complication in this ISO-based procedure is that assignment of these numbers (3 incandescent lamps, 2 fluorescent lamps, etc.) depends on how the process data are specified and is therefore part of the Inventory analysis; thus, it cannot take place in the prior Goal and scope definition step.

In this Guide the concept of reference flow has therefore been adapted somewhat. The use processes and quantified flows are specified in the Inventory analysis. Goal and scope definition is concerned with a different issue: in this case, the specification that 1000 hours of light with an incandescent lamp are to be compared with 1000 hours of light with a fluorescent lamp. These two items then constitute the reference flows for the two systems. They form two alternative ways of supplying the functional unit (1000 hours of light). These alternatives, and thus these two reference flows, may be referred to as 'products' in the generalised sense in which a barber or teacher supplies a product. In this sense, ISO's definition of a reference flow ("the amount of product which is necessary to fulfill the function")[1] can even be retained. Summarising, reference flows are product-specific flows having the functional unit as a common denominator:

- reference flow for system 1: 1000 hours of light with an incandescent lamp;
- reference flow for system 2: 1000 hours of light with a fluorescent lamp;
- functional unit: 1000 hours of light;
- process data (3 incandescent lamps, 2 fluorescent lamps, 200 kWh electricity, etc.): to be specified in the step 'Data collection and relating data to unit processes', in the Inventory analysis phase.

One interesting advantage of this modification is that it allows behavioural differences between the alternative systems to be explicitly incorporated in the specification of the reference flows. Tube lights, for instance, tend to be switched off less frequently than other lamps. The function 'lighting a kitchen for 365 evenings' may then well involve 3000 hours of tube light versus 1000 hours of light with a different lamp, with subsequent explicit consequences for the reference flows. The consequences further on in the Inventory analysis relate to well-defined flows for different amounts of light and electricity.

Based on the above reasoning we come to the following definitions for the functional unit and the reference flow (adapted from ISO 14040):

- Functional unit: quantified service provided by the product system(s) under study for use as a reference basis in a life cycle assessment study.
- Reference flow: quantified flow generally associated with the use phase of a product system and representing one way (i.e. by a specific product alternative) of obtaining the functional unit.

Given these adapted definitions, the discussion on actual, recommended and standard performance (and related issues) is an issue connected with the step 'Data collection and relating data to unit processes', in the Inventory analysis. Guidelines on this subject are therefore given in Part 2a, Section 3.3.6.

With respect to reporting, Lindfors *et al.* (1995a) list the following issues for the step Function, functional unit, alternatives and reference flows:

- The main user function(s) forming the basis for the LCA shall be clearly defined and reported.
- The functional unit(s) of the studied system(s) shall properly describe the services provided and be clearly defined and reported.
- A brief description of the product group under study (functions) is recommended.
- The studied product or service alternatives shall be described and the choice justified.
- Relevant alternatives not covered by the study should be commented upon or listed in the report. If no relevant alternative exists, it is equally important to make such a statement.

Meier *et al.* (1997) give the following reporting guideline:

- The functional unit of the system under study should be unambiguously defined and should reflect the actual function of the system in a measurable and quantitative way. The functional unit should be relevant to the goal of the study.

PROSPECTS

No specific developments are foreseen in this area.

CONCLUSIONS

Concluding, we recommend following the stepwise structure proposed in ISO/TR 14049 (1998) with a minor addition:

[1] Note that this definition of reference flow, on p. 5 of ISO 14041 (1998), differs from the definition on p. 2 of the same document.

Identification of functions
identify all relevant functions of the product systems studied.

Selection of one or more functions as relevant
- select one or more functions as the relevant functions for comparison;
- if more than one function is relevant:
  - take into account the primary function only and neglect *all* other functions, or
  - take into account the primary function and (all) the additional functions, or
  - allocate between the primary function and the additional functions, selecting the appropriate option for the particular goal of the study.

Defining the functional unit
- specify selected function(s) in relevant SI or SI-derived units;
- determine an arbitrary or absolute (e.g. annual) quantity.

Determining equivalent alternative (product) systems and reference flows
determine and specify the alternative systems studied in terms of the key parameters of the reference flow (e.g. trip rate, life span, mass and volume material specifications, etc).

All four steps are relevant for comparative studies and comparative assertions. In non-comparative studies the first step is optional and may be omitted, since only one system is then considered regardless of the functions it fulfils.

RESEARCH RECOMMENDATIONS
Short-term research
- How to deal with the divergence between standard, recommended and actual use.

Long-term research
- Incorporating monetarily defined functional units in the LCA framework.
- Describing 'lifestyles' in terms of operational functional units.

3. Inventory analysis

3.1 General introduction

In the Goal and scope definition phase of an LCA the basic groundplan for the study is established, as discussed in Chapter 2. The second phase of an LCA is the Inventory analysis (see Figure 3.1.1).

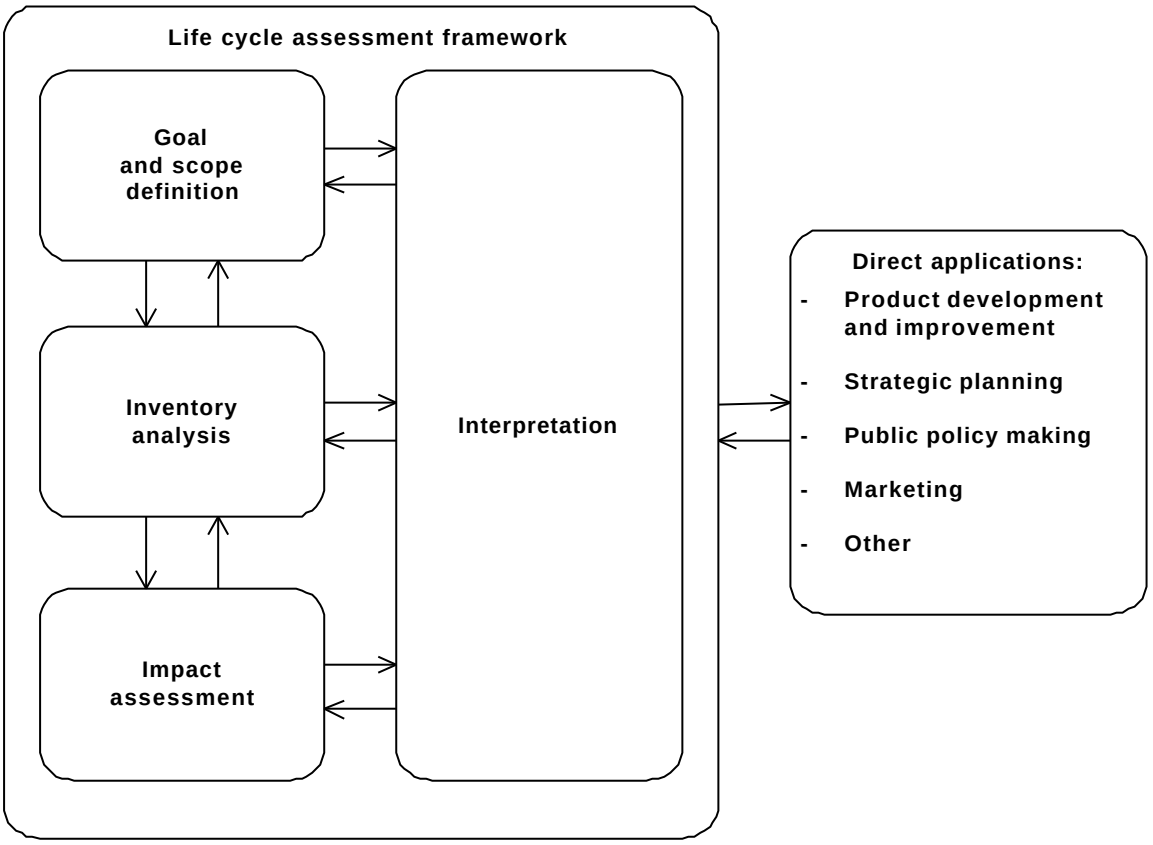


Figure 3.1.1: The Inventory analysis phase as part of the general methodological framework of LCA (source: ISO 14040, 1997E).

According to ISO 14041 (1998E) this phase is concerned with "the collection of the data necessary to meet the goals of the defined study" and with the associated "data collection and calculation procedures" and "is essentially an inventory of input/output data with respect to the system being studied." The aim is thus to prepare an inventory, first qualitative then quantitative,of all the processes involved in the life cycle of the product (or function) system(s) under study (herafter: 'product system'), detailing all relevant interactions with the environment. The Inventory analysis is generally the most time and resource consuming phase of an LCA.

The product system, defined by ISO 14041 (1998E) as "a collection of unit processes connected by flows of intermediate products which performs one or more defined functions", affects the environment via environmental interventions such as resource extractions or emissions of hazardous substances. The basic element of the Inventory analysis is thus the unit process. The term unit process refers to any kind of activity producing an economically valuable output such as steel, electricity or bread, or providing an economically valuable service such as transport or waste management. Products, services or resources, etc. are converted into other products, services and emissions. Every product and service used is fed by

other unit processes and every product and service produced (particularly waste to be processed) feeds in to other processes.[1]

These supply and consumption processes are by definition included as part of the product system when the complete life cycle is considered. Proceeding from the reference flow specified in the goal and scope phase (see Figure 2.4.1), upstream and downstream unit processes are identified based on the economic in- and outflows of a process. The first process is usually a use-related process, e.g. use of a refrigerator, watching television, etc. The processes collected in this way together constitute the life cycle of the product system investigated. If more product systems are studied in a given study, these operations are repeated for each system.

The processes in the life cycle can be arranged in the form of a flow diagram. A flow diagram is a systematic arrangement of the main processes which make up the product system. In drawing the flow diagram, first the boundary between the product system and the environment system needs to be determined. For example, it has to be decided whether landfill and agricultural production are to be included as unit processes within the system boundary. Next, the characteristics and data of each process included in the flow diagram have to be determined. Some processes and their characteristics are available in publicly accessible databases or literature. Nevertheless, many data will have to be collected in dedicated literature studies. Process modeling (e.g. process engineering models) may also be useful. Particular attention should be given to the quality of the data collected, by comparison with data from other, similar sources and by compiling mass and energy balances, for example.

Next, the problem of processes supplying multiple functions needs to be addressed. The production of chlorine, sodium hydroxide and hydrogen by electrolysis of common salt is an example of a multiple process. Since these products are produced in fixed quantities and since product systems do not generally need both products, at least not precisely in the fixed quantities, an allocation step is required. In this allocation step the multiple process is often divided into two or more single processes, purely for analytical purposes.

Finally, the quantity of each single function required by the product system is calculated for each single process. All products and materials are balanced by multiplying each process by the appropriate number: the volume produced is exactly that required in the subsequent process and all that remains at a systems level is a service which is provided externally, that is, the functional unit. Their aggregation across all the processes produces the inventory table.

The general operating procedure for the Inventory analysis as briefly introduced above is the one generally encountered in LCA software programs. To be complete, however, the Inventory analysis should include a number of additional steps, among them preparatory steps, to be performed before any data is entered or software calculations made, and several steps relating to procedures, reporting and other ‘miscellaneous’ issues.

As elaborated in this Guide, the Inventory analysis therefore distinguishes ten steps (see Section 1.4):

- Procedures (no special section in this Part; see Chapter 1);

Compilation of flow diagram and process data

- Economy - environment system boundary (Section 3.2, p. 87);
- Flow diagram (Section 3.3, p. 91);
- Format and data categories (Section 3.4, p. 93);
- Data quality (Section 3.5, p. 99);
- Data collection and relating data to unit processes (Section 3.6, p. 104);
- Data validation (Section 3.7, p. 109);
- Cut-off (Section 3.8, p. 109);

Allocation

- Multifunctionality and allocation (Section 3.9, p. 116);

Calculation of inventory results

- Calculation method (Section 3.10, p. 134).

Further points of departure in elaborating these Inventory analysis steps are ISO documents 14040 (1997E) and particularly ISO 14041 (1998E) with respect to the methodological framework and the issues for Inventory analysis proposed there. Below, the ISO proposals are further operationalised, taking into account the work of the SETAC Working Group on Inventory enhancement (Clift *et al.*, 1998) and relevant proposals by other authors. Deviations from ISO arise only if there are significant reasons for doing so.

[1] For brevity and readability, we shall often write ‘process’ instead of ‘unit process’.

All steps distinguished above will be discussed according to a fixed format: 'Topic', 'Developments in the last decade', 'Prospects', 'Conclusions' and 'Research recommendations'. As explained in Section 1.5 the step dealing with procedures is not discussed separately in this chapter, but in integrated fashion, for all phases, in Section 1.3.

It should be noted once again that our focus here is the change-oriented type of LCA. Moreover, the steps elaborated below assume a number of simplifications in the modeling set-up, as discussed earlier in Section 1.2 of this Part of the Guide. The key simplifications introduced are: use of linear relations and exclusion of most economic, socio-cultural and technological mechanisms. Thus, fixed input/output relations and a 'technical' production function are employed. In this sense the inventory model specified here is a mechanical one.

3.2 Economy - environment system boundary

TOPIC

In LCA each and every flow should be followed until its economic inputs and outputs have all been translated into environmental interventions. The term 'environmental interventions' refers to flows entering the (product) system (natural resources, for example, but also land use) which have been drawn from the environment without prior human transformation, or flows of materials leaving the (product) system which are discarded into the environment without subsequent human transformation (Figure 3.2.1). Environmental interventions are thus flows crossing the boundary between the economy (= product system) and the environment. To create a clear distinction between the product system and the environment and between elementary and other flows, the economy-environment boundary should be explicitly defined.

Whenever a system is studied, system boundaries are needed to separate the system from the rest of the world. In LCA Inventory analysis three types of boundaries can be distinguished (Heijungs *et al.*, 1992):

1. the boundary between the product system and the environment system;
2. the boundary between processes that are relevant and irrelevant to the product system (cut-off);
3. the boundary between the product system under consideration and other product systems (allocation).

The first of these will be discussed here, the second under the heading 'Cut-off and data estimation' (Section 3.8) and the third under the heading 'Multifunctionality and allocation' (Section 3.9).

Although some boundaries of the first type are very clear, there are major areas of ambiguity with regard to the boundary between the product and environment system, as is illustrated by the cases of landfill sites and agricultural soil. These issues of ambiguity will be examined below.

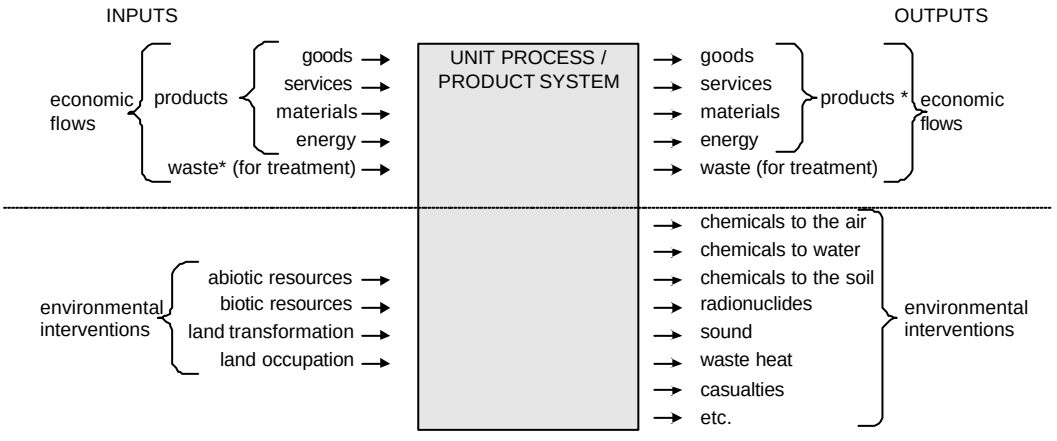


Figure 3.2.1: Environmental interventions and economic flows.

DEVELOPMENTS IN THE LAST DECADE

The issue of system boundary definition is treated in two different sections of ISO 14041:
in 'Scope of the study': 5.3.3 Initial System boundaries, as part of the Goal and scope definition; in 'Calculation procedures': 6.4.5 Refining the system boundaries, as part of the Inventory analysis.
The definition of the boundary between the product system and the environment system is

Heijungs *et al.* (1992)
The system boundary between the product system and the environment system is explicitly mentioned in the 1992 guide (pp. 27–28):
- a distinction is made between economic and environmental processes. It is stated that almost any activity incurring costs is an economic process;
- treatment steps occurring after a substance has been introduced into the environment (e.g. purification of river water to produce drinking water) should not be included in the product system causing the emission;
- landfilling of waste is to be considered an economic process;
- processes relating to agriculture, livestock management, forestry, etc. are to be considered economic processes;
- in processes involving personnel, consideration might be given to including additional physiological and economic processes (increased metabolism, eating and drinking, commuting etc.).

implied in the statement "Ideally, the product system should be modeled in such a manner that inputs and outputs at [its] boundary are elementary flows" (Section 5.3.3). An elementary flow is defined in ISO 14040 (1997E) as "material or energy entering the system being studied, which has been drawn from the environment without previous human transformation; material or energy leaving the system being studied, which is discarded into the environment without subsequent human transformation". ISO thus makes an explicit statement here as to how this boundary should be defined; the key distinction is "human transformation".
In defining the product system-environment boundary there is no consideration for the 'grey' area ambiguity, however, with questions such as "does the agricultural topsoil belong to the environment or to the product system studied" remaining unanswered.

In the Nordic Guidelines (Lindfors *et al.*,1995a,b), the boundaries between the Technosphere and the Biosphere are defined similarly to the boundaries between the product system and the environment system in Heijungs *et al.* (1992). The authors mention that "any change in natural production rate caused by a technical activity should thus be accounted for". Furthermore, "natural biomass production as well as emissions from landfills should in principle be included in the inventory, whenever relevant". According to these authors the time horizon for emissions from landfill should be set by the temporal boundary defined in the Scope definition.
The report of the Dutch Platform 'LCA & Waste' (Udo de Haes *et al.*, 1997) describes the results of discussions on modeling end-of-life processes in LCA among a group of some 30 Dutch LCA experts between November 1995 and September 1996. The group of experts concluded that landfill sites could either be regarded as part of the product system or as part of the environment system. The choice depends on the degree of control at the site: a very well controlled, i.e. managed, site might be regarded as part of the product system, an uncontrolled site as part of the environment system.
We conclude with respect to landfill sites that a useful distinction can be made between controlled and uncontrolled sites. The latter should be regarded as part of the environment system, with flows to them regarded entirely as emissions, while controlled sites form part of the product/function system. Actual estimation of landfill emissions is covered in Section 3.6 on data collection. Emissions from controlled landfills should be quantified to an infinite time horizon in detailed LCA studies. However, the actual duration of relevant emissions will be far shorter than eternity, depending on the types of wastes involved and landfill site conditions. Following an initial period, all wastes exhibit a tendency towards reduced emission activity or, conversely, towards increased immobilisation. For household wastes, for example, a time horizon of 200 years seems to cover virtually all emissions, in temperate climates, as far as organic substances are concerned. In very dry and hot regions, this time horizon may be much longer. In the very long run, as considered in steady-state modeling, leaching will remove virtually all inorganics such as heavy metals. For organic chemicals, emissions from landfill should therefore be integrated over 200 years and for inorganic chemicals, over infinity. The latter means that each input of inorganic chemicals to landfill is regarded as an emission to soil. As with fate modeling, it is questionable whether extremely long time horizons are entirely relevant for interpretation with respect to inorganics. Hence, it

may be useful to perform a sensitivity analysis on emissions over the first 200 years only ('cut-off') or to discount them in some, as yet unspecified, way (*cf.* Finnveden, 1999b).

With respect to the economy-environment boundary in agriculture there have been several developments have in the past decade. The report 'Application of LCA to Agricultural Products' (Wegener Sleeswijk *et al*., 1996) focused on LCAs for agricultural products. One of the subjects discussed in this report was the boundary between the product system and the environment. There it was opted to include the agricultural soil in the environment system, for the main reason that damage to the soil should be regarded as an environmental impact in order to differentiate between systems differing in their impact on soil quality[1]. Furthermore, it was opted to regard the harvested portion of the crop as an economic output of arable farming and thus as part of the economic system, with the remaining portion being regarded as part of the environment system. A consequence of this choice is that use of pesticides is regarded as an emission to the environment, except for pesticides which end up on the harvested crop. Horticultural production in which no natural soil is used for production belongs entirely to the economy, except for the soil itself, which remains part of the environment system. Another argument for defining the soil as part of the environment stems from the principle of 'multifunctionality'. This principle, regularly applied in the context of public policy, implies that the quality of, say, agricultural soil should be maintained at such a level that it can also fulfil other functions, including ecological functions. If land is taken out of agricultural use, the quality of the soil should be such as to permit other types of land use. Other choices are possible here. Audsley *et al*. (1994), for example, opted to regard soil as part of the economy, right down to the depth of the water table, because soil is an integral part of farming systems. In specific agricultural studies, the analysis of the top layer may be of importance. In general LCA a simple system boundary excluding soil seems adequate enough.

In this Guide the multifunctionality principle is followed. Thus, agriculture and forestry are taken to be economic processes, agricultural and forestry soils remain part of the environment system, the harvested portion of the crop flows to other processes in the economic system while the non-harvested portion remains in the environment system.

System boundary problems may also occur when pollutants are moved from one environmental compartment to another, as is the case when contaminated sediments from surface waters are transferred to adjacent land, i.e. soil (Gorree & Kleijn, 1996; Blonk & Van Ewijk, 1996) when waterways and harbours are dredged. Since no 'new' pollutants are added to the environment (no new emissions take place), the effects would not be visible in an LCA. In the report of Gorree & Kleijn, however, it is proposed that sediment removal from surface waters be treated in the same way as CO_2 uptake by production forests: as a negative emission (see textbox). The distribution of this sediment on the surrounding soil is subsequently treated as a 'positive' emission to the terrestrial environment.

> Carbon dioxide, as a naturally occurring compound or an anthropogenic emission, takes part in the so-called geochemical carbon cycle. Short and long carbon cycles can be distinguished. Reforestry, for example, is a process with a short carbon cycle: during tree growth a certain amount of atmospheric CO_2 is fixed, but is ultimately released (as CO_2 or CH_4) when the wood is landfilled, incinerated or decays naturally. The most appropriate way to treat short carbon cycles is to view them as cycles and thus, at the systems level, subtract the fixation of CO_2 during tree growth from the CO_2 emitted during waste treatment of discarded wood and to quantify the CH_4 emitted (this balancing of carbon as an element in compounds such as CO_2 and CH_4 is necessary because CO_2 and CH_4 have different characterisation factors with respect to global warming ($GWP_{CO2,100} \div GWP_{CH4,100} = 1 \div 11$)). A detailed elaboration of this kind of short carbon cycle has been developed by Virtanen and Nillson (1993) in their study on paper board.
>
> An example of a long carbon cycle is geochemical carbon fixation in fossil fuels and atmospheric release of carbon in the form of CO_2 and CH_4 when these fuels are burned. With long carbon cycles, carbon fixation is too slow a process and CO_2 and CH_4 emissions should be accounted for in their entirety, without further balancing against prior fixation.
>
> Source: Guinée (1995).

Concluding, in agricultural LCAs a distinction between 'negative' and 'positive' emissions may be relevant. In every LCA, moreover, it is very important to ensure that system boundaries are defined

[1] An exception here is horticultural production, where production generally takes place on an artificial substrate rather than natural soil. Such production belongs entirely to the economy.

consistently throughout, i.e. with respect to characterisation factors and normalisation factors (cf. Section 4.6).

A final issue of possible ambiguity with respect to system boundaries concerns waste water treatment plants (WWTP). Bearing in mind the definition of unit process, a WWTP should be regarded as an economic process. This implies that releases to a sewage system are not considered as an emission into the environment, but as a flow to the unit process WWTP. Only the releases of treated waste water from the WWTP to surface (fresh) water are taken to be emissions. Because WWTP process data are sometimes difficult to obtain, in practice waste water treatment will sometimes be omitted from the flow diagram. In such cases this should be clearly reported and justified, however.

Clear and detailed reporting of choices, data and assumptions is crucial for the entire Inventory analysis. This is particularly important with respect to the definition of the economy-environment boundary, since sound decision-making demands clarity on what has been included in the environment and what not. General reporting guidance is provided by a number of documents.
The general requirements of ISO 14040, clause 6, cited in Chapter 2, also apply here. In addition, ISO 14041 (1998E), clause 8 gives the following requirements for a third-party study report on Inventory analysis:
"The results of an LCI study shall be fairly, completely and accurately reported to the intended audience as described by the relevant parts of clause 6 of ISO 14040 (1997E). If a third-party report is required, it shall cover all items marked with an asterisk. All additional items should be considered."
More specifically with respect to system boundaries, we have the following recommendations by Lindfors *et al*. (1995a):

- The initial system boundaries chosen in relation to the goal and scope of the study shall be reported.
- The detailed system boundaries, viz. geographical, life-cycle and technosphere-biosphere boundaries, shall be described and justified in full.

The SETAC-Europe Case studies Working Group (Meier *et al*., 1997) give the following (minimum) reporting guidelines:

- Systems and system boundaries should be clearly defined for all stages of the product system life cycle, including inputs, processing routes, spatial and temporal considerations. Ancillary data for the product system should be clearly defined. Exclusions should be stated and justified along with a consideration of the significance of any exclusions on the outcome of the study.

PROSPECTS

If some degree of differentiation as to the time of impacts is to be included in the Impact assessment phase, this will have consequences for the Inventory analysis in terms of how the relevant time horizon is to be specified. How this is to be combined with steady-state modeling is a conceptually complex question. No other developments are foreseen for this topic.

CONCLUSIONS

Concluding, with respect to some of the principal ambiguities regarding the boundary between the economy and the environment we recommend the following:

agricultural production
soil and unharvested parts are allocated to the environment

forestry
soil and unharvested parts are allocated to the environment

natural forests
growth of natural forests is not part of the economy

wild fish, game, fruits, herbs
growth is not part of the economy

positive/negative emissions

The distinction between positive and negative emissions is relevant for chemicals absorbed from the environment or consituting an emission from a human activity, such as dredging or agriculture. Dredging for specific purposes (shipping channels, clay for bricks or dykes) should be included as a negative emission (from the environment) , and as a positive emission when returned to the environment (particularly when this is to a different environmental compartment). In agriculture, and particularly in forestry (*note* : not natural forests), sequestering of CO_2 in biomass should be considered a negative emission, while CO_2 or CH_4 released during waste processing of agricultural products should be taken as a positive emission.

landfill emissions
for controlled landfills, emissions of organics should be integrated over 200 years, emissions of inorganics *ad infinitum*; for uncontrolled landfills, all inputs of potentially hazardous chemicals should be considered as immediate emissions to the environment. As with fate modeling, it is questionable whether extremely long time horizons are entirely relevant for interpretation with respect to inorganics. Hence, it may be useful to perform a sensitivity analysis on emissions over the first 200 years only ('cut-off') or to discount for them in some, as yet unspecified, way.

mining tips
to be treated in the same way as landfill

waste water treatment plants
WWTPs should be regarded as an economic process, implying that only the releases of treated waste water from the WWTP to surface (fresh) water are considered as emissions. Because WWTP process data are sometimes difficult to obtain, in practice waste water treatment will sometimes be omitted from the flow diagram. In such cases this should be clearly reported and justified, however.

In the case of landfill and mining tips, when boundaries are still unclear we recommend adopting a pragmatic approach. Mining wastes disposed of in mines can be regarded as returning to the lithosphere and do not then constitute emissions (provided the mine is still as isolated from its biotic surroundings as it was before it was opened). Regulated use of waste materials on soil substrates, as when mining wastes are applied under permit for the foundation of roads or buildings, do not constitute emissions either, although subsequent emissions from the new road structure should be duly recorded. At controlled landfill sites, only the emissions out of the site are recorded, in the way specified above. At non-regulated landfill sites, inflowing wastes are also treated in their entirety as emissions. For partially controlled sites, a mix may be applied.
Finally, there is the question of how to treat short and long carbon cycles. Short carbon cycles should preferably be regarded as cycles and thus, at the systems level, the fixation of CO_2 during tree growth should be subtracted from the CO_2 emitted during waste treatment of discarded wood and any CH_4 emissions should be quantified. For long carbon cycles, CO_2 and CH_4 emissions should be recorded in their entirety, without further balancing against prior fixation.

RESEARCH RECOMMENDATIONS
- Elaborate the specification of temporal aspects in a steady-state modeling context, especially for landfill and long-cycle systems, *inter alia* in relation to temporal aspects of Impact assessment.
- Detail the boundaries between Technosphere, Biosphere and Lithosphere in a more systematic manner.

3.3 Flow diagram

TOPIC

The flow diagram provides an outline of all the major unit processes to be modeled, including their interrelationships. It is helpful in understanding and completing a system to describe the system using a process flow diagram.

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Heijungs *et al.* (1992)
The life cycle consists of interlinked economic processes, each process input coming either from another process or directly from the environment and each process output flowing either to another process or to the environment, at least after allocation. Economic processes are taken to comprise resource extraction, production of materials and components, product manufacture, product use and associated waste processing, including recycling and reuse, and a variety of ancillary processes, such as transport and electricity generation. Any processes omitted should also be specified.
In practice a summary flow diagram will first be compiled that includes only the key processes of resource extraction, product manufacture and associated transport. The interconnected processes underlying each of these are then represented separately in partial flow diagrams, allowing the practitioner to 'zoom in' to the items of interest.

With respect to the topic of flow diagram ISO states only that "it is helpful to describe the system using a process flow diagram showing the unit processes and their interrelationships". In addition to this initial flow diagram, preparation of 'specific' flow diagrams is discussed as the first step in the paragraph on Preparing for data collection (6.2): "drawing of specific process flow diagrams that outline all unit processes to be modeled, including interrelationships".
Drawing up an initial flow diagram can be regarded as a practical aid to data collection. As it will only become apparent which economic flows and delivering processes are relevant to a particular study as data are retrieved, compiling a definitive flow diagram will be a highly iterative process. Initially, the diagram will cover only the process delivering the functional unit and the adjacent processes supplying the main raw materials and treating the principal waste flows, and their respective interconnections. Even after all the relevant data have been collected, though, a truly comprehensive diagram will in practice be impossible to compile. Given the common occurrence of process loops and multi-outputs, such a diagram would simply include too many unit processes in overly complex interrelationship. In most cases, then, the only workable solution is to draw up a basic flow diagram showing the main constituent unit processes, from which to 'zoom in' to the underlying unit processes, represented separately in partial flow diagrams of their own.

Wegener Sleeswijk & Huppes (1994) propose the following conventions for constructing flow diagrams.
- unit processes are represented as boxes;
- economic flows are represented as arrows between such boxes;
- economic flows enter a unit process at the top of the box and leave a unit process at the bottom of the box;
- the main direction of flow through the flow diagram is top-down, although recursive flows are in the reverse direction;
- all boxes contain text labels with the name of the process, e.g. 'production of sulphuric acid';
- arrows are labeled only in so far as their name is required for an unambiguous understanding of the flow diagram;
- a flow diagram should preferably not contain more than 20 boxes;
- use of partial flow diagrams may be useful; for each box, it should then be clearly indicated whether it represents an undivided unit process or an aggregated unit process, possibly elaborated elsewhere in a partial flow diagram.

These rules should be seen as recommendations for an optimum use of flow diagrams rather than as binding guidelines.

For reporting flow diagrams Lindfors *et al.* (1995a) provide the following recommendations:

- The studied systems should be defined and reported using flow diagrams at the maximum level of detail used in the study.
- Sub-systems may be aggregated to higher levels if appropriate, as long as detailed descriptions are provided (e.g. in an appendix).

On this issue Meier *et al.* (1997) recommend that the process flow diagrams reported describe the entire system under study and include system boundaries, major inputs, products and co-products, as well as the main production sequence, ancillary materials and energy/fuel production.

PROSPECTS

No specific developments are foreseen for this topic.

CONCLUSIONS

Concluding, we recommend:

- to distinguish between an initial flow diagram, at the level of aggregated processes for each life cycle stage, and a detailed flow diagram, at the level of (possibly) unit processes;
- to denote (aggregated) processes by boxes and economic flows by arrows and adopt the other recommendations of Wegener Sleeswijk & Huppes (1994);
- to exclude environmental flows from flow diagrams, for pragmatic reasons; and
- to draft the flow diagram or diagrams as an iterative process during the data collection step.

RESEARCH RECOMMENDATIONS

No specific research is recommended.

3.4 Format and data categories

TOPIC

A key task of the Inventory phase is the collection of process data. This usually involves large quantities of data in electronic form, retrieved in part from databases set up by others. To render these comparable and mutually consistent a standard *data format* must be developed. All the various data categories should be assigned a specific place in this format and a general description given of each to facilitate and guide data entry and retrieval.

Ideally, the data formats used for data exchange (paper version of technical software format) and for (software) processing should be identical. However, as software packages differ in terms of underlying data model (often unspecified), no overall format for data processing can be recommended. Such a format can be drawn up for data exchange, however, reducing substantially the efforts required for processing with specific software.

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Although ISO 14041 (1998E) does not distinguish a separate 'format' step, clause 5.3.4 covers 'Description of data categories' which is clearly related. This, in Annex A4 of ISO 14041 (1998E), an example is given of a Data Sheet for a Unit Process. This Annex is provided for purely illustrative purposes. In the accompanying text it is stressed that there are no fixed rules for either the number of data categories or the amount of additional information required on the quality and uncertainty margins of the data. The format given in the example has entry spaces for material, energy and water inputs, material outputs (including products) and emissions to air, water and land as well as other releases (e.g. noise, radiation, vibration, odour, waste heat, etc.). There is also space for reporting other characteristic data on the process in question.

On the topic of data categories ISO 14041 (1998E) clause 4.4 states, furthermore, that:
"the major headings under which data can be classified include:

- energy inputs, raw material inputs, ancillary inputs, other physical inputs;
- products;
- emissions to air, emissions to water, emissions to land, other environmental aspects.

Heijungs *et al.* (1992)

In the '92 guide the data categories are discussed in 2.2.1 *Quantification of the inputs and outputs*. It is stated that since all economic outputs of processes are connected to economic inputs of other processes these input and output categories should be fully symmetrical. For both inputs and outputs a distinction is made between goods, services, materials, energy and waste for treatment. Environmental inputs and outputs are not symmetrical. In the '92 guide the following inputs from the environment were distinguished: abiotic resources, biotic resources, energy resources and space. The environmental outputs were as follows: emissions to air, water and soil, radiation, noise, heat, light and accidents. It is also mentioned that some environmental outputs may be of a more qualitative nature, such as ecosystem fragmentation due to road-building programmes. "Negative emissions" are also mentioned, occurring mainly in processes on the boundary between the environment and the economy, e.g. the uptake (negative emission) of CO_2 by forests and crops.

Section 2.2.1 of the '92 guide also introduces the term *format* in relation to the manner in which LCA Inventory data are to be stored and processed. A distinction is made between the *conceptual format* and the *technical* (i.e. software) *format*. The conceptual format relates to the main structure (see Figure 3.2.1), the technical format to the rules for filling in the process data. The main structure indicates the input and output categories of the process: economic inputs and outputs, and environmental inputs and outputs (see Figure 3.2.1) and also Appendix A.1. of Heijungs *et al*., 1992). The '92 guide does not elaborate the technical format in any further detail (although CML later drew up a such a format based on the conceptual format).

Besides the economic and environmental inputs and outputs, the conceptual format also includes the following aspects:

- who entered the data, and when
- representativeness of the data (scale, dating, duration, status)
- quality of the data (clarity, accuracy, completeness)
- source of the data
- mass and energy balance of the process.

There is also reference to the fact that unquantifiable aspects may be important for some processes. The conceptual format explicitly provides space for this purpose.

The background to the '92 guide (section 2.3.1) specifies a number of general requirements the format should meet:

Methodological delineation:

- The format will have to be particularly oriented towards those environmental interventions ultimately of importance for Classification.
- The format will have to be oriented toward the "potentially relevant" economic inputs and outputs.

Other requirements:

- The format shall be geared to practical data availability.
- The format shall be in line with current practice wherever possible
- The format shall be readily comprehensible and suitable for international use.
- The format shall be such as to prevent LCAs being swamped by having to fill in an excessive number of process data.

Within these headings, individual data categories shall be further detailed to satisfy the goal of the study. For example, under emissions to air, data categories such as carbon monoxide, carbon dioxide, sulfur oxides, nitrogen oxides, etc. can be separately identified."

In clause 5.3.4 ISO 14041 continues: "The data required for an LCA study are dependent on the goal of the study. Such data may be collected from the production sites associated with the unit processes within the systems boundaries, or they may be obtained or calculated from published sources. In practice, all data categories may include a mixture of measured, calculated or estimated data. Subclause 4.4 [see above] outlines the major headings for the inputs and outputs that are quantified for each unit process within the systems boundary. These data categories should be considered when deciding which data categories are used in the study. The individual data categories should be further detailed to satisfy the goal of the study.

Energy inputs and outputs shall be treated as any other input or output to an LCA[1]. The various types of energy inputs and outputs shall include inputs and outputs relevant for the production and delivery of fuels, feedstock energy and process energy used within the system being modeled.

[1] In many contexts there is a special interest in the extraction of energy resources. As the depletion of energy resources is treated in the Impact assessment as part of general abiotic depletion, energy resource use may be specified in terms of the total energy content of all the abiotic energy resources involved, e.g. using the heat

Emissions to air, water and land represent discharges from point or diffuse sources, after passing through emissions control devices. The category should also include, when significant, fugitive emissions. Indicator parameters, e.g. biochemical oxygen demand (BOD), may also be used.
Other data categories for which input and output data may be collected include, for example, noise and vibration, land use, radiation, odour and waste heat."
As mentioned, ISO 14041 has little to say on data format as such. It is nonetheless a very important issue, particularly when it comes to data exchange. Streamlining of data exchange can lead to substantial savings on both the time and resources required for an LCA study.
A format is characterised by:

1. its conceptual, i.e. primary structure;
2. the items it comprises: the spaces to be filled in by the user;
3. the rules for filling the spaces: 'the cookbook'.

When drawing up a format, due allowance should be made for the fact that various different kinds of process data will be employed in any given LCA.
Raw process data (from a variety of sources with their own data formats) can be processed by various procedures into four basic forms of process data;

1. unallocated unit process data;
2. allocated unit process data;
3. (partial) system data retraceable to unit process data
4. (partial) system data irreversibly aggregated.

In principle, these various kinds of data set have different format requirements. In the case of 4. partial system data irreversibly aggregated, for example, it is necessary to indicate the allocation procedures and cut-offs employed and so on, while with unallocated unit processes these issues have no relevance.
Another important point is the distinction between data and system. More particularly, the question is then *whether* and, if so, *how* the various processes (of whatever category) are interconnected *within the database*. A database may, for example, comprise data on a whole series of processes stored completely independently of one another. It may also contain data on process interlinkages, however, in which case there is in fact no longer any distinction between data and system. The way in which the processes are interlinked then forms the system; *cf.* 3, above, "system data retraceable to process data".
In the LCA context, data and system should preferably not be combined until such time as an actual case study is to be performed. An LCA database may sometimes contain two different data sets for the same or comparable processes, for example when two literature sources report different data for the same process, or when there are two ways of producing a given material. In each specific case study one particular process, one particular literature source or one particular production method will have to be selected. What occurs in a case study is thus a stringing together of a series of selected processes. All this has consequences for the data format. Where a given process input goes to or where a given output comes from does not therefore belong in the process data format but in the description of the case study in question. In short, then, data and system should in principle be kept separate. This is seldom the case in current LCA databases, however.

The format itself, a user data sheet with a certain structure, is in principle an entirely separate issue from how the data are stored. One of the standard methods for data storage is in a relational database (as in SPINE and in Huber, 1996). The essential feature of a relational database is that all independent information is stored at a single location, which has the major advantage that updates need only be introduced at this one location. This is achieved by the data for a given process being stored in a great many different tables. This concept forms the basis of virtually all commercial database software, such as DBASE, ORACLE and MS-ACCESS, and has for years been the standard method for storing large amounts of data. Use of a relational database demands high standards of database integrity and the same holds for all the various elements of the user data sheet. Current data exchange formats like SPOLD have opted for easy readability and are not relational in design. When updating such non-relational databases severe inconsistencies are to be expected.
If use is made of a relational database it would seem an obvious approach to employ it to store data not only data on individual unit processes but also on process interlinkages. As already mentioned, the latter kind of data relate to the system rather than to individual processes. As such, then, they do not

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values as specified in the ETH database on energy transformation processes. This information does not enter the computations in the Impact assessment, however. [Addition from authors of this Guide.]

belong in a discussion of the format for data on unit processes. Tables with data relating to Classification/Characteriserion and Evaluation may also be included. These data should also be independent of the Inventory data.

Since 1992 a number of different data formats for LCA have been developed around the world, most of them in the form of tabular data sheets. Some, such as SPOLD and SPINE (discussed below) have been implemented in database software, while others are available as a printed form only. A number of LCA databases have also been developed that store process data in a proprietary format (e.g. ETH, SimaPro, SAEFL). Here we restrict ourselves to a discussion of SPINE and SPOLD.

SPINE

SPINE, the Sustainable Product Information Network for the Environment, is a joint initiative of the Swedish organisation IVL and the Chalmers University of Technology established in the context of the Nordic project on Environmentally sound Product development (NEP). The SPINE report (Steen *et al*., 1995), has been adopted by ISO as a "New Work Item Proposal" and is to be further elaborated. Although the proposal names SPOLD (see below) as a 'Liaison Organisation', the precise mutual status of these two formats is not yet entirely clear. At the time of writing of the present Guide, the outcome of the work based on this proposal was not known.

SPINE is a relational database designed to permit communication between different software tools. The key question addressed during development of SPINE was: what kind of information is of value for LCA and how are the various data related? The key concept of the database is that of an *activity*, to be interpreted here as either a unit process or a (partial) system. An activity has quantifiable inputs and outputs. An important aspect of SPINE is that it stores not only actual the process data but also data on the flows between the various activities, thus creating a network-type model. The SPINE model is linear and homogeneous. The database has extensive capacity for storing qualitative information on all elements (flows, activities, substances, etc). This qualitative information is stored in separate tables, essentially independent of the quantitative data, and of little significance for the actual functioning of the database. SPINE can be used to store not only inventory data but also Impact assessment data (equivalency factors and so on), although the designers note that the latter module is still under development. (The subject of 'Multifunctionality and allocation' is treated in Section 3.9 of this Guide.) Processes can be aggregated to 'mega-processes' by means of a hierarchical structure. Data input to SPINE is by means of a screen form, similar to the aforementioned printed forms, and is indeed also available as a printed form. It is to be concluded that SPINE has a solid, intrinsically consistent format. One potential drawback of SPINE is the rigidity resulting from using the relational database concept.

SPOLD

SPOLD, the Society for the Promotion Of Life-cycle assessment Development, seeks as one of its missions to establish a comprehensive database of commodity materials and services. An important aspect of SPOLD's work is the key focus on achieving consensus. In 1996 a SPOLD Status Report was published entitled 'Introduction into a Common Format for Life-Cycle Inventory Data' (Singhofen, 1996; *cf.* Singhofen *et al.,* 1996). In contrast to SPINE, for example, this document describes a paper format and distinguishes between a descriptive and a prescriptive approach. Given the importance attached to consensus by SPOLD, it was ultimately opted to start with a descriptive approach, providing as much scope as possible for using existing data(bases) and including all information deemed relevant. An important feature of the SPOLD format is that it has space for data on both unit processes and partial systems (aggregated unit processes). Thus the first question users are asked during data input is whether the data relates to a unit process or a partial system. Although in theory information should obviously be collected at the unit process level, in practice data are often found to relate to (partial) systems and SPOLD consequently reserves due space for the latter.

The SPOLD format has five components:

A. identification of data set (who, what, how?)
B. system description, in cases involving system data (what is included, what assumptions?)
C. graphic presentation of the system (now abandoned by SPOLD)
D. input and output data, plus space for mass and energy balances
E. references.

Part A: Identification of data set

- do the data relate to a unit process or a partial system (aggregated unit processes)?

- who entered the data, and when?
- geographical, temporal and technological representativeness
- sources

Part B: System model
- subsystems
- cut-off criteria
- coproduction and associated allocation rules
- energy, transport and waste model
- other assumptions

Part C: System structure
- graphic presentation of the model (abandoned by SPOLD for mainly practical reasons)

Part D: Inputs en outputs
- Inputs
  - from the technosphere (economy): materials/fuels, electricity/heat
  - from nature (environment): natural resources
  - other
- Outputs
  - to the technosphere (economy): (co-)products, waste
  - to nature (environment): air, water, land, non-material
  - other interventions
- Mass and energy balances

Part E: References

The aim of SPOLD is to use the aforementioned descriptive format to develop a second-generation format, this time prescriptive. The new format is intended for use with a dictionary (list of definitions) and a multi-user test, leading to an electronic format that can serve as a basis for a data exchange network. In the meantime the first-generation (descriptive) SPOLD format has also been made available electronically, downloadable from Internet.
Goedkoop (1998) has developed a simplified version of the original SPOLD format, the main feature of which is that a large number of specific fields have been replaced by 'free' text fields with no fixed structure. This is most relevant in part B of the SPOLD format: system data (as opposed to unit process data). One benefit of the simplified format is that it is more user-friendly for LCA practitioners, while still dovetailing well with the original SPOLD format. A disadvantage is that use of a greater number of free text fields means it is no longer relatively straightforward to export data to the full SPOLD format.

DALCA
DALCA (Van Dam *et al.*, 1996), implemented by TNO, stands for *DAta for Life Cycle Analysis*. DALCA is a "feasibility study on the future availability of universally accepted, reliable environmental data on processes (as 'building blocks' for environmentally oriented product information)". DALCA focused on products manufactured by the plastics processing industry. A data format is presented in Appendix 2 of the report. It is largely in line with that provided in the '92 guide and consists of four tables:
1. data identification
2. data
3. transport
4. verification.

The first table covers process identification, including process description, representativeness, author(s) and flow diagram. The second is for average data, with space provided for data margins and quality. The third provides dedicated space for transport data. The fourth and last table is a proposal for presentation of process verification and includes such aspects as: comparison with external sources, completeness, internal consistency (incl. balances), reliability, status, data gaps and so on. The principal difference from the format of the '92 guide is the specific focus on transport and verification. It is noteworthy that the verification table asks for an explicit indication of who is responsible for verification: the company, an independent institute, the subcontractor or others.

EDIP (Wenzel *et al.*, 1997) stands for Environmental Design of Industrial Products and has been developed by the Institute for Product Development, the Technological University of Denmark, five Danish industries, the Confederation of Danish Industries and the Danish EPA. Wenzel *et al.* 1997 briefly describe a paper format for use in LCAs. Although it is largely similar to the format of the '92 guide, one important difference is the attention given to characterising the geographical site of the process in question and the extent to which the data are specific to this site. There is even space for including location-specific data relating to characterisation, normalisation and evaluation. The format also has space for information on working conditions.
Boguski (1996) provides a review of LCA, including inventory and data format. Chapter 2 of this work gives a basic data format that is again largely similar to that of the '92 guide, differing in the following main respects: less focus on qualitative information such as representativeness, and subcategorisation of economic inputs and outputs into transport, off-site energy purchases, autogenerated power and coproducts.
As part of the Dutch MRPI project ('Environmentally Relevant Product Information for the Construction Industry') a format has been developed for collecting data on processes such as those occurring in the construction sector (Anonymous, 1998). MRPI is concerned with "validated information on the environmental aspects of building materials, building products or building elements, drawn up at the initiative of the producer or his representative (the trade association, for example) by means of an environmental life cycle analysis" (Anonymous, 1998). The process data required for the LCA are collected together on a printed form. This format, however, is devoted specifically to the construction sector and building materials. Although the design of this form is too specific to serve as a general LCA format, it might serve as a useful aid in drawing up such a format.

Principal differences
Each of the formats discussed above provides space for quantitative information on the economic and environmental inputs and outputs of the process under study. SPINE is the only format based on the relational database concept, which is a solid guarantee of database integrity. A key feature of SPINE is that it stores not only the actual process data but also provides space for including process interlinkages, i.e. system data. It is, of course, debatable whether this format is 'complete' and whether all the items included really belong there. Another important respect in which various formats differ is in the space provided for qualitative information. This varies from single-line entry (the name of the process) to an extended list of items relating to data quality and representativeness and sometimes to (partial) system data such as allocation methods used, cut-off criteria and so on. They also differ in the space provided for indicating data error margins: some formats ignore this issue entirely, while others provide scope for including upper and lower bounds or spread for all data entered. In addition, most of the formats traced are paper formats, although there are plans to convert some of them to electronic form. Finally, the formats differ in the degree to which they break down economic and environmental inputs and outputs into such categories as transport, energy and materials: some formats have many such subcategories, while others have none.

PROSPECTS
Although there is presently no ISO standard for data format, a Swedish application to develop a format on the basis of Spine and SPOLD was recently honoured by ISO. As the authors themselves state, the current version of the SPOLD form is based on a descriptive analysis of the formats currently used by LCA practitioners. This means that the format is not entirely homogeneous and that some of the items included are questionable. A second-generation SPOLD form is to be developed, based on a prescriptive format.
With respect to data format a choice must be made from among the following options:
1. one of the existing formats
2. a modified version of an existing format
3. a combination of several existing formats
4. an entirely new format.

The overriding aim in designing a data format is to guide LCA users in collecting process data, and not to establish how these data are to be stored and coupled. The new Guide therefore includes a data sheet rather than a description of underlying data structure requirements. We would stress once again, though, that a data structure in the form of a relational database is a solid guarantee of database integrity.

On the basis of the aforementioned and other criteria, a variety of forums including the SETAC Working Group on Data availability have expressed a preference for the SPOLD user form

CONCLUSIONS

The data sheet that is best supported by existing software is the SPOLD form. It is consequently recommended to use this form as long as there is no easy-to-implement alternative employing a relational database. It is to be noted that SPOLD and SPINE are meanwhile being combined by way of ISO/TR 14048 (in prep.). These developments, which had just begun when this Guide was being written, should therefore be followed closely. Once these efforts yield a practicable form(at), this should be used instead of the SPOLD format.

RESEARCH RECOMMENDATIONS

- Research should focus on use of relational databases in LCA and on translating relational database requirements into a data format and data sheet. Relational databases appear to be a very promising tool for ensuring the integrity of the LCA database and, consequently, that of the LCA results. Especially promising in this respect is the initiative, in the ISO context, to combine SPOLD en SPINE: see ISO/TR 14048 (in prep.).

3.5 Data quality

TOPIC

For LCA models, like any other model, it holds that 'garbage in = garbage out'. In other words, data quality has a major influence on results and proper evaluation of data quality is therefore an important step in every LCA. Even if the quality of individual data is high, however, such data can still yield erroneous results if used to answer questions on which they have limited or no bearing. The data used in a given case study should, for instance, be representative for that particular study. Quality requirements thus refer to both the reliability and the validity of process data. As validity depends on the application in question, it is not validity requirements as such that are specified here but the data needed to assess that validity.

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)

The '92 guide treats this topic in section 2.2.2, on the representativeness and quality of data, as part of the Inventory analysis (pp. 32–33). The following sub-steps are distinguished:

- the representativeness of the processes;
- the quality of the process data;
- the overall assessment of the process data.

In ISO 14041 (1998E), clause 5.3.6, where 'Data quality' is part of the 'Scope of the study', the following statements are made with regard to data quality issues (see textbox):

Descriptions of data quality are important for understanding the reliability of the results of a study and properly interpreting its outcome. Data quality requirements shall be defined to enable the goal and scope of the study to be met. Data quality should be characterised in both quantitative and qualitative terms and with due reference to the methods used to collect and integrate the data.

Data quality requirements should be set for the following parameters:

– temporal coverage: the desired age of data (e.g. within the last five years) and the minimum period of time (e.g. annual) over which data should be collected;
– geographical coverage: geographical area from which data on unit processes should be collected to satisfy the goal of the study (e.g. local, regional, national, continental, global);
– technology coverage: technology mix (e.g. weighted average of the actual process mix, best available technology, or worst operating unit).

Consideration shall also be given to additional descriptors defining the sort of data required, e.g. collected from specific sites versus data from published sources, and whether the data is to be measured, calculated or estimated. Data from specific sites or representative averages should be used for those unit processes contributing the majority of the mass and energy flows in the systems under study, as determined in the sensitivity analysis performed in 5.3.5. Data from specific sites should also be used for unit processes deemed to have environmentally relevant emissions.

In all studies, the following additional data quality requirements shall be considered at an appropriate level of detail depending on the Goal and scope definition:

– precision: measure of the variability of the data values for each data category expressed (e.g. variance);
– completeness: percentage of locations reporting primary data relative to the potential number in existence for each data category in a unit process;
– representativeness: qualitative assessment of the degree to which the data set reflects the true population of interest (i.e. geographical, temporal and technology coverage);
– consistency: qualitative assessment of how uniformly the study methodology is applied to the various components of the analysis;
– reproducibility: qualitative assessment of the extent to which information about the methodology and data values allows an independent practitioner to reproduce the results reported in the study.
– Where a study is used to support a comparative assertion that is disclosed to the public, all data quality requirements described in this subclause shall be included in the study.

Where a study is used to support a comparative assertion that is disclosed to the public, all data quality requirements described in this subclause shall be included in the study.

Source: ISO 14041, 1998E.

The aforementioned EDIP report (Wenzel *et al.*, 1997) emphasises the importance of process characterisation. Data quality is described at two levels: the level of individual inputs and outputs and that of the whole process. According to EDIP, data characterisation should generally cover the following items:

– Definition of the scope of the process[1]
  - description of operations included and excluded and of inputs and outputs not linked to other economic processes
  - specification of co-products produced and method for allocating these
– Data characterisation
  - description of known data gaps
  - description of the data source
  - how well do the data describe the process and how representative of the average are they?
  - description of the representativeness of the process for the objective of the study
  - description of the assessment/calculation of the coefficients of statistical variation for the environmental inputs and outputs
  - mass balance: calculation of the mass balance for the process
– technological development
  - description of technological developments and trends in the most important inputs and outputs
  - description of the projection of the process[2]

[1] In the EDIP report the format is described in both table 22.5 and figure 9.1, which unfortunately use different titles and terminology. Here, the terminology from table 22.5 has been employed.

[2] The term 'projection' in the EDIP book refers to the extrapolation of process data to the future year in which the newly developed product is to be launched on the market.

Several remarks are in order. In the assessment of statistical variation, uncertainties in economic flows may be more important than those in environmental inputs and outputs. Furthermore, it may be useful to make a clear distinction between the overall validity and reliability of data (and other data quality aspects) at the process level, independent of the application involved (as given by database requirements, for example), and their validity and reliability in a specific application.

Nordic Technical Report No. 9 (Lindfors *et al.*, 1995b) starts out by referring to data collection as the most time-consuming and thus the most expensive element of LCA. In screening product LCAs the amount of time required for this task can be reduced by using readily accessible data. However, this data may be of inferior quality. The report mentions three aspects to which due attention should be paid when selecting data for use:

- level of technology;
- age of data;
- site specific vs. average data.

Lindfors *et al.* (1995b) consider all three aspects very important for the results of the study. A number of parameters are suggested for assessing the quality of databases used in LCA:

- correctness;
- reliability;
- integrity;
- usability;
- portability;
- maintainability;
- flexibility;
- testability.

Building on Weidema (1994), Lindfors *et al.* (1995a) elaborate a detailed scheme of data quality parameters, represented as a 'data pedigree matrix' (Table 3.5.1). They propose reporting data in the following format:

2.3 MJ - 0.1, %95 (2,2,3,1,1,1)

A B C D (a,b,c,d,e,f)

where

A = the magnitude of the data as a numerical or descriptive expression (here: 2.3)

B = the 'unit base operation' as a standard SI-unit and multiplier (here: Mega, 10^6)

C = the spread of the data as a percentage range or standard deviation (here: - 0.1)

D = an assessment of the reliability of data, confidence interval or qualitative description (here: %95)

and

a,b,c,d,e,f represent the origin (pedigree) of the data and thus its representativeness (see Table 3.5.1 below).

Table 3.5.1: Pedigree matrix suggested by Weidema (1994) for formalised presentation of LCA data (source: Lindfors *et al.*, 1995a).

position	**a**	**b**	**c**	**d**	**e**	**f**
pedigree	acquisition method	independence of data supplier	representativeness of sample	data age	geographical correlation	technical correlation
1	measured data	verified information from public authority or other independent source	data from continuous measurements over an adequate period at a sufficient sample of enterprises to even out normal fluctuations	recent (maximum 3 years)	data from area under study	data from enterprise under study
2	calculated data based on measurements	verified information from enterprise with interest in the study	sample data or data from continuous measurements at a smaller number of enterprises but over an adequate period	less than 5 years	average data from larger area that includes area under study	data on same processes/materials but from different enterprises
3	calculated data based partly on assumptions	independent source but based on non-verified information from industry	data for shorter period but from continuous measurements at a sufficient sample of enterprises	less than 10 years	data from area with similar production conditions	data on same processes/materials but with different technology
4	qualified estimate (by industrial expert)	non-verified information from industry	sample data for shorter period but from a sufficient sample of enterprises	less than 20 years	data from area with slightly similar production conditions	data on similar processes/materials, with similar technology
5	non-qualified estimate	non-verified information from enterprise with interest in the study	representativeness unknown, or single or sample data from one enterprise over a shorter period	age unknown, or more than 20 years	data from unknown area or area with very different production conditions	data on similar processes, materials but with different technology

Weidema (1998b), building on Wesnaes & Weidema (1996), presents an adapted version of the pedigree matrix set up by Lindfors *et al.* (1995a). According to the author "the objective of the pedigree matrix is to provide a data quality management tool, which makes it easy to survey the data quality, to point at possibilities for improvements in data quality and to trace back sources of uncertainty".

Table 3.5.2: : Adapted version of the data pedigree matrix (source: Weidema, 1998b).

pedigree	**reliability**	**completeness**	**temporal correlation**	**geographical correlation**	**further technical correlation**
1	verified data based on measured data	representative data from a sufficient sample of sites over an adequate period to even out normal fluctuations	less than 3 years of difference from the year of study	data from area under study	data from enterprises, processes and materials under study
2	verified data based partly on assumptions, or non-verified data based on measurements	representative data from a smaller number of sites but for adequate periods	less than 6 years of difference	average data from larger area that includes area under study	data on processes and materials under study but from different enterprises
3	non-verified data partly based on assumptions	representative data from an adequate number of sites but for shorter periods	less than 10 years of difference	data from area with similar production conditions	data on processes and materials under study but with different technology
4	qualified estimate (e.g. by industrial expert)	representative data but from a smaller number of sites, for shorter periods, or incomplete data for an adequate number of sites and periods	less than 15 years of difference	data from area with slightly similar production conditions	data on related processes or materials but with same technology
5	non-qualified estimate	representativeness unknown, or incomplete data from a smaller number of sites and/or for shorter periods	age of data unknown, or more than 15 years of difference	data from unknown area or area with very different production conditions	data on related processes materials but with different technology

It is not, as yet, feasible to reason backwards from a required level of confidence in the results of an LCA study to specification of requirements vis-à-vis the validity and reliability of the model and data to be employed. Nor is there yet a comprehensive method for establishing the overall quality of results on the basis of quantitative yardsticks of reliability and validity. Full statistical analysis, including analysis of error propagation, will not be available to LCA practitioners for a long time yet, if ever. The most one can do is trace back the factors determining the overall quality of the results, and make some semi-quantitative elements operational to aggregate the otherwise very substantial amount of data on quality aspects that accumulate in a study. A framework for this has been elaborated by Van den Berg *et al.* (1999) with some preliminary operationalisations. The main philosophy followed is that of Funtowicz & Ravetz (1990). For some elements of this framework semi-quantitative approaches are available and may be applied (Weidema, 1998b; Van Oorschot, 1999; Van den Berg *et al.*, 1999)

In relation to reporting, Lindfors *et al.* (1995a) suggests that if risk of accident is signalled with a 'red flag', the criteria for such a flag shall be reported. They state, furthermore, that "data and data quality issues shall be reported giving:

1. Information on the source of data used
2. At least a qualitative declaration of data quality (specific or average, from what year, representativeness, estimate of variability (uncertainty). If not known, this should be stated."

Meier *et al.*, 1997 give the following reporting guideline: "all data sources for the study should be clearly identified and referenced. Key indicators of data quality should be reported, such that choices made are justified and transparent, including

1. age of data
2. frequency of collection
3. spatial/temporal considerations
4. accuracy
5. precision
6. uncertainty
7. etc.

If data are presented in an aggregate form (i.e. for confidentiality reasons) the aggregation procedure should be fully described. A sensitivity analysis should also be performed on key data sets."

PROSPECTS

Significant progress may be expected in this area in the coming years, but a full traditional error analysis will probably never be possible in LCA since 1) there is no normal (gaussian) distribution, as processes are highly non-linear; and 2) there is also no normal distribution for the inter-process connecting flows.

CONCLUSIONS

Based on the above analysis of the state-of-the-art in the field of data quality assessment in LCA, we recommend as the best available practice for today:

- to follow the ISO 14041 (1998E) requirements (5.3.6) concerning data quality and optionally use a 'pedigree matrix' for reporting data quality assessments on individual processes;
- to trace back the factors determining the overall quality of the outcome by undertaking a Contribution analysis (see Section 5.4) and a Perturbation analysis (see Section 5.5) , and to perform a sensitivity analysis on a number of key parameters (see Section 5.6); and
- (optionally) to apply a general framework for quality assessment of results (e.g. Van den Berg *et al.*, 1999), and to apply semi-quantitative approaches, as available, for some of the elements of such a framework (e.g. Weidema, 1998b; Van Oorschot, 1999; Van den Berg *et al.*, 1999).

Given this unsatisfactory state of affairs it is very important not only to present the results of the study in a transparent way (a quality aspect itself) but also to perform a quality assessment and report on it in an explicit and transparent manner, mentioning the desired as well as minimum data quality goals sought.

RESEARCH RECOMMENDATIONS

- It is not yet possible in LCA to elaborate a standardised method for overall assessment of data quality. For the process data used in the Inventory phase this would, for example, mean having to know the probability interval of every input and output of every process involved in the life cycle. Similarly, the validity of the inventory model and the environmental effect models would have to confirmed. At the moment appropriate yardsticks of statistical distribution and a validation procedure to quantitatively assess overall model validity are both lacking.

3.6 Data collection and relating data to unit processes

TOPIC

In this step of the Inventory analysis all relevant data on the unit processes are collected and all flows connected to the unit processes quantified in accordance with the format specified above (Section 3.4). The reference flow(s) (see Figure 2.4.1) defined in the Goal and scope definition phase of the study form(s) the point of departure for data collection. The process data available to the practitioner may be structured in any number of ways. In LCA databases process data is often organised around unit processes, relating a given economic output to economic inputs and environmental inputs and outputs. Process data provided by companies are often also organised around unit processes, but given in terms of inputs and outputs per unit time, e.g. emission of 5 tonnes of CO_2 per year, input of 1000 tonnes of wood per year, etc. In current LCA databases process data is almost always quantified in relation to some physical (reference) flow (e.g. one kg of material or 1 MJ of electricity). In ISO 14041 (1998E) this step of relating all inputs and outputs to one reference flow is described in paragraph 6.4.3 "Relating data to unit processes".

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)
The step *Entering the process data* in the '92 guide is more or less equivalent to ISO's data collection step. The former was divided into two sub-steps (pp. 30–35):
1) quantification of the inputs and outputs (including format and quantification of data)
2) representativeness and quality of the data.
The second topic has been discussed here in the previous section (see section 3.5). Under the first heading several topics were considered. The central topic, data format and data categories, is discussed here in section 3.4. Other items discussed included the non-linear characteristics of most economic processes. The solution adopted for this problem there was to work with long-term marginal process data. Here, this discussion has been further refined and can be found in section 1.2.3.4 and section 2.3. Furthermore, it was recommended to use SI units whenever possible.

In ISO 14041 (1998E) three steps appear to be distinguished:
1) preparing for data collection
2) data collection
3) validation of data
These are mainly procedural steps. On preparing for data collection, ISO 14041 (1998E) states the following (see text box):

The definition of the scope of an LCA study establishes the initial set of the unit processes and associated data categories. Since data collection may span several reporting locations and published references, several steps are helpful to ensure uniform and consistent understanding of the product systems to be modeled.
These steps should include:
- drawing of specific process flow diagrams that outline all unit processes to be modeled, including interrelationships;
- description of each unit process in detail and listing of data categories associated with each unit process;
- development of a list that specifies the units of measurement;
- description of data collection techniques and calculation techniques for each data category, to assist personnel at the reporting locations to understand what information is needed for the LCA study; and
- provision of instructions to reporting locations to document clearly any special cases, irregularities or other items associated with the data provided.

An example of a data collection sheet is provided in annex A.

Source: ISO 14041, 1998E.

With regard to data collection, ISO 14041 (1998E) states that "the procedures used for data collection vary with each unit process in the different systems modeled by an LCA study. Procedures may also vary due to the composition and qualification of the participants in the study and the need to satisfy both proprietary and confidentiality requirements. Such procedures and reasons should be documented."

In clause 5.3.3 on initial system boundaries and clause 6.3 on data collection, ISO 14041 states (see text box):

> Each of the unit processes should be initially described to define:
> - where the unit process begins, in terms of the receipt of raw materials or intermediate products;
> - the nature of the transformations and operations that occur as part of the unit process; and
> - where the unit process ends, in terms of the destination of the intermediate or final products.
>
> Data collection requires thorough knowledge about each unit processes. To avoid double counting or gaps, the description of each unit process shall be recorded. This involves a quantitative and qualitative description of the inputs and outputs which are needed to determine where the process starts and ends, and the function of the unit process. Where the unit process has multiple inputs (e.g. multiple effluent streams to a water treatment plant) or multiple outputs, data relevant for allocation procedures shall be documented and reported.
>
> When data are collected from published literature, the source shall be specified. For those data collected from literature which are significant for the conclusions of the study, the published literature which supplies details about the relevant data collection process, about the time when data have been collected and about further data quality indicators, shall be specified. If such data do not meet the initial data quality requirements, this shall be stated.
>
> Source: ISO 14041, 1998E.

On the subject of relating data to unit processes, ISO 14041 (1998E) clause 6.4.3 states: "For each unit process, an appropriate reference flow shall be determined (e.g. 1 kg of material or 1 MJ of energy). The quantitative input and output data of the unit process shall be calculated in relation to this reference flow."[1]

The EDIP books (Wenzel *et al.*, 1997*)* explicitly mention the importance of a data check when electronic databases are used and distinguish several types of processes, based on the possible suppliers of such data:

- raw materials extraction and materials production: producers' associations and private companies;
- product manufacturing processes and use processes: private companies;
- disposal processes: public authorities, research institutes;
- transport processes and energy systems: specialized information centres;

Wenzel *et al.* (1997), furthermore, mention the future need for so-called *data networks* in contrast to current *static databases*. In these data networks each individual expert body must handle and update its part of the database on the product system. They mention, as a precondition for ensuring viability, that each body has an intrinsic commercial interest in participating.

Another interesting item in the EDIP report is that they provide some basic guidelines for collecting data on the working environment. In this context four different impact categories are mentioned:

- chemical impacts;
- impacts of noise;
- impacts of monotonous repetitive work;
- risk of accident.

Wegener Sleeswijk *et al.* (1996) mention several issues specific to Inventory analysis in LCAs on agricultural products. Such LCAs involve a number of problems connected with the production level to which the data to be gathered should relate. Compared with many other economic activities, agricultural production encompasses a relatively large number of production units (in this case: farms). The production processes - and consequently the associated environmental interventions, too - may differ markedly from farm to farm. Therefore the results of the LCA will be dependent on the type of data used: average or representative data, or data from individual farms. The type of data to be chosen depends on the Goal and scope of the study. For example, if the aim is to inform consumers about the environmental impacts of a certain product, such as a bottle of milk, one should, in principle, review the particular farm from which the milk in the given bottle originates. In practice, however, the bottle will contain milk from different farms, mixed at large production centres, and average data should therefore

[1] Note that this 'reference flow' is different from that defined and used in section 2.4 on Function, functional unit, alternatives and reference flows

be used to describe the environmental impacts of the product. If a government is aiming to assess the variation in impact of current modes of milk production, data from individual operational farms should be used. If, on the other hand, the goal is to assess future scenarios based on current policy trends, use should be made of normative data (representative for modern farms).
In principle this problem is not restricted to agricultural LCAs but may extend to all LCA studies. An interesting case in point is the modeling of waste management processes for long-lived products. In a study on water pipes, crash barriers and roof gutters by Kortman *et al.* (1996) waste management scenarios were developed for the year 2015.

For communication reasons, it may be useful to make a distinction here between primary data (measured on-site) and secondary data (data from literature and databases, for example, or estimates from IOA models). Additionally, a distinction is often made between so-called *foreground systems* and *background systems* (SETAC Clift *et al.*, 1998; see also Anonymous, 1999). Clift *et al*. (1998) defined the foreground system as the set of processes whose selection or mode of operation is affected directly by decisions based on the study. The background system comprises all other processes interacting directly with the foreground system. It was stressed by Clift *et al*. (1998) that the distinction between foreground and background systems (or processes) has nothing to do with the environmental importance of their respective impacts; either the foreground or the background system may have the greater impact (Clift *et al*., 1998). For foreground processes primary data will normally be collected, while for background processes use will generally be made of secondary data sources.
As Clift *et al*. (1998) state, the distinction between foreground and background will frequently be clear, for example, when the decision-maker is the operator of a set of processes constituting the natural foreground. In other cases it will not be as clear, however, for example when LCA is used as a basis for purchasing decisions. Background processes may then be defined as those processes underlying a market seen as more or less homogeneous from the perspective of the process providing the functional unit. In order to avoid these rather theoretical discussions, a more pragmatic definition will be used in this Guide:

- foreground systems or processes are those systems or processes for which primary, site-specific data are used in an LCA (for whatever reason);
- background systems or processes are those systems or processes for which secondary data from databases, public references or estimated data based on IOA models are used.

The more foreground processes a specific LCA study includes, the more ‘detailed’ the LCA will be.
In LCA practice the use of generic databases is almost indispensable for background processes. There are several databases available, differing in terms of status, spatial, temporal and technical representativeness, types of processes covered and data formats employed. It should be duly checked whether the unit process data in the database has already been subjected to allocation procedures. LCA practitioners should be aware that a choice for a particular database may greatly influence the ultimate results of the study (e.g. Copius Peereboom *et al*., 1998).

Due attention should also be given to the units in which the collected data are expressed, to ensure that all the processes can be mathematically connected once all the data is available.
With respect to units, there are three issues to which particular attention should be paid:

- conversion of Bq to kg, and vice versa;
- conversion of dB to $Pa^2 \times yr$; and
- calculation of ‘land occupation’ and ‘land transformation’ as data subcategories of ‘land use’.

Practical tips are provided for all three issues in Part 2b, Section 3.6.

One special issue in data collection is how data relate to unit processes. As Curran (1996) states, "raw materials and energy data for production facilities are often expressed in terms of annual or monthly production”. These numbers will therefore have to be translated to units per quantity of product. This presupposes a separate step translating time-related data to data per quantity of product (reference flow), which then form the input for LCA databases or case studies. For unallocated databases this involves the arbitrary choice of making one of the products into a reference flow. The need for such translation is debatable: it is an extra step that can be easily avoided by using a scaling factor that includes time as a dimension (Heijungs, 1998a). One can even argue that valuable information on the magnitude of a process is lost in the process of translation. Data from a number of important data sources such as corporate environmental plans and reports, emission registration systems and statistical information systems often include the time dimension. If these data are transferred to (a)

database(s) with no prior translation, the same database(s) can be used for LCA and for other environmental tools like Substance Flow Analysis (SFA), Material Flow Accounting (MFA), Risk Assessment (RA), etc. However, LCA software tools do not always permit use of process data that include a time dimension. Omitting this dimension in LCA calculations poses no problem, however, as it is the ratio between inputs and outputs that is used here. Since there are strong arguments for having access to process data that include a time dimension and because most LCA data currently consist of data related to a reference flow, it is strongly recommended to develop LCA software and databases that allow both types of process data to be used (for an example of operational software that includes this option, cf. http://www.leidenuniv.nl/interfac/cml/ssp/cmlca.html).

Finally, the issue of group or sum parameters needs to be discussed. Some chemicals, such as PAHs and CFCs, are recorded in the Inventory analysis as group parameters[1], although characterisation factors are only available for individual chemical species, such as anthracene and chrysene, or CFC–11 and CFC–12. Other group parameters regularly encountered in inventories for which this problem occurs include Volatile Organic Compounds (VOC), sulphur compounds, absorbable organic halogens (AOX) and hydrocarbons (CxHy). These group parameters should preferably be broken down into their individual chemical constituents and specified as such. If data on the 'real' constituents is not available, group parameters should be broken down into their individual chemical constituents using generic conversion factors like those published by Derwent *et al*. (1996) or used for Dutch Emission Registration. If these options are not feasible or do not cover the group of chemicals in question, one should take the arithmetic mean of the individual species of the group as a surrogate characterisation factor. If emission data are available for 'hydrocarbons' only, for example, a group POCP can be derived as the arithmetic mean of the POCPs of the individual hydrocarbons. This approach is a method of last resort and, in this example, is only of value if the POCPs of the individual hydrocarbons vary within a 'reasonable' range. Because of the often large number of emissions encompassed in 'organic compounds' or 'AOX', however, and the major spread in characterisation factors, use of an arithmetic mean would introduce unacceptable uncertainties. In such cases a 'best estimate' should be used to specify the group parameter in terms of consistuent species for the purpose of characterisation. Although incomplete, any specification is better than none and also generally superior to an 'arithmetic mean' approach.

PROSPECTS

No specific developments are foreseen for this topic.

CONCLUSIONS

ISO 14041 requirements with respect to the description of unit processes, data categories and data collection procedures, etc. have been adopted in this Guide.

Furthermore, it seems useful for communication reasons:

- to distinguish between primary data (measured on-site) and secondary data (from literature and databases, for example, or estimates from IOA models); and
- to distinguish between:
  - foreground systems or processes: systems or processes for which primary, site-specific data are used (for whatever reason);
  - background systems or processes: systems or processes for which secondary data are used.

The more foreground processes a specific LCA study includes, the more 'detailed' the LCA will be.

Finally, SI-based or -derived units should be used throughout and unit process data should preferably be collected in terms of annual flows in order to allow use of these data by other environmental tools.

Note 1: as discussed in Section 2.4 on 'Function, functional unit, alternatives and reference flows', the issue of standard, recommended and actual performance should be dealt with here, although the discussion of developments on this topic was treated in Section 2.4. The recommendation with respect to this subject is: quantify the key parameters of the system's reference flow, preferably based on:

- actual performance, and otherwise
- standard or recommended performance.

Note 2: the issue of technology coverage has been discussed in Section 2.3 on 'Scope definition', where it is recommended to take current state-of-the-art technology as a starting point for data collection.

[1] Group parameters are valid parameters only if measured as such, not when calculated from measurements on individual chemicals.

RESEARCH RECOMMENDATIONS

– Since data availability is one of the most important problems in LCA practice, it is recommended to establish a project to construct a database comprising a set of reference processes with their corresponding interventions. Use of the Internet as a medium for sharing LCA data should also be developed further. Furthermore, LCA software should preferably allow scaling of data, including a time dimension.

3.7 Data validation

TOPIC

In this step the validity of the process data collected is checked. Various tools are available for this purpose, including mass balances, energy balances and comparison with data from other sources (e.g. comparative analysis of emission factors). Any data found to be inadequate during the validation process should be replaced. Similarly, missing data should be identified in this step and a decision made on how these gaps are to be filled.

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* 1992

Validation of process data is partly discussed in section 2.2.2 of Heijungs *et al.* "The representativeness and quality of the data". In the discussion on "The quality of the process data" verification of the data by means of mass and energy balances is mentioned, as well as a check on data completeness.

With respect to 'data validation' ISO 14041 (1998E) states the following (see textbox):

A check on data validity shall be conducted during the process of data collection. Validation may involve establishing, for example, mass balances, energy balances and/or comparative analysis of emission factors. Obvious anomalies in the data appearing from such validation procedures require alternative data values which comply with the data quality requirements as established according to 5.3.6.

For each data category and for each reporting location where missing data are identified, the treatment of the missing data and data gaps should result in:

– a data value which is justified;
– a "zero" data value if justified; or
– a calculated value based on the reported values from unit processes employing similar technology.

The treatment of missing data shall be documented.

Source: ISO 14041 (1998E).

Although the general topics of data quality and data quality indicators are discussed in a number of publications, including Wenzel *et al*. (1997) and Lindfors *et al*. (1995a), the topic of data validation as described here is not found in the literature except in ISO 14041 (1998E), being subsumed elsewhere under general data quality assessment.

PROSPECTS

No specific developments are foreseen for this topic.

CONCLUSIONS

Concluding, we recommend following the ISO 14041 requirement to check the validity of the process data collected by drafting mass balances and energy balances, by comparison with data from other sources (e.g. comparative analysis of emission factors).

RESEARCH RECOMMENDATIONS

No specific research is recommended.

3.8 Cut-off and data estimation

TOPIC

In principle an LCA should track all the processes in the life cycle of a given product system, from the cradle to the grave. In practice this is impossible, however, and a number of flows[1] must be either roughly estimated or cut off[2] and subsequently ignored. The root problem behind the cut-off issue is a lack of readily accessible data, implying disproportionate expenditure of funds and effort on data collection. Cut-off may substantially influence the outcome of an LCA study, however, and means that 'easy' LCAs come at a price. The cut-off criteria specified in the past, such as omitting mass flows under 1 (or 5)%, lead to the fallacy of disaggregation: by splitting up processes after a more detailed review, most flows can be reduced to less than the specified percentage. If the rule is that a cut-off can be introduced if contributions to impact assessment results are below a certain percentage, there is a danger of the same fallacy occurring. In addition, it seems odd to omit data from the computations having gone to all the effort of generating them. Thus, the cut-off problem can be reformulated as a problem of having to quantitatively estimate the environmental interventions associated with flows for which no readily accessible data are available.

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* 1992

The '92 guide states the following with regard to the boundary between relevant and irrelevant processes:

- a boundary should be drawn somewhere to avoid the problem of *infinite regression;*
- it should be decided whether such items as capital goods or a staff canteen should be included in a process;
- the preferred criterion for excluding particular processes is a quantitative estimate of their relative contribution to environmental effects;
- the costs of maintenance and depreciation may provide another indicator; if these form a substantial part of the product price, the environmental intervention associated with capital goods should not be excluded *a priori.*

In ISO 14041 (1998E) the subject of cut-off is given ample attention. In clause 5.3.3 ISO 14041 states the following with respect to the definition of initial system boundaries (see text box):

[1] In the cut-off discussion, the term "flows" refers specifically to all economic input flows and the output flow "waste to be treated".

[2] In section 2.3, on scope definition, one situation was noted in which a cut-off may be introduced on grounds other than those discussed below: a difference analysis.

The systems boundary defines the unit processes which will be included in the system to be modeled. Ideally, the product system should be modeled in such a manner that inputs and outputs at its boundary are elementary flows. In many cases there will not be sufficient time, data, or resources to conduct such a comprehensive study. Decisions shall be made regarding which unit processes will be modeled by the study and the level of detail to which these unit processes shall be studied. Resources need not be expended on the quantification of such inputs and outputs that will not significantly change the overall conclusions of the study.

Decisions shall also be made regarding which releases to the environment shall be evaluated and the level of detail of this evaluation. In many instances those system boundaries defined initially will subsequently be refined on the basis of the outcome of the preliminary work (see 6.4.5). The decision rules used to assist in the choice of inputs and outputs should be clearly understood and described. Further guidance on this process is provided in 5.3.5.

Any decisions to omit life cycle stages, processes or data needs shall be clearly stated and The systems boundary defines the unit processes which will be included in the system to be modeled. Ideally, the product system should be modeled in such a manner that inputs and outputs at its boundary are elementary flows. In many cases there will not be sufficient time, data, or resources to conduct such a comprehensive study. Decisions shall be made regarding which unit processes will be modeled by the study and the level of detail to which these unit processes shall be studied. Resources need not be expended on the quantification of such inputs and outputs that will not significantly change the overall conclusions of the study.

Decisions shall also be made regarding which releases to the environment shall be evaluated and the level of detail of this evaluation. In many instances those system boundaries defined initially will subsequently be refined on the basis of the outcome of the preliminary work (see 6.4.5). The decision rules used to assist in the choice of inputs and outputs should be clearly understood and described. Further guidance on this process is provided in 5.3.5.

Any decisions to omit life cycle stages, processes or inputs/outputs shall be clearly stated and justified. The criteria used in setting the system boundaries dictate the degree of confidence in ensuring that the results of the study have not been compromised and that the goal of a given study will be met.

Several life cycle stages, unit processes and flows should be taken into consideration, e.g.:

- inputs and outputs in the main manufacturing/processing sequence;
- distribution/transportation;
- production and use of fuels, electricity and heat;
- use and maintenance of products;
- disposal of process wastes and products;
- recovery of used products (including reuse, recycling and energy recovery);
- manufacture of ancillary materials;
- manufacture, maintenance and decommissioning of capital equipment;
- additional operations such as lighting and heating;
- other considerations related to Impact assessment (if any).

It is helpful to describe the system using a process flow diagram showing the unit processes and their interrelationships. Each of the unit processes should be initially described to define:

- where the unit process begins, in terms of the receipt of raw materials or intermediate products;
- the nature of the transformations and operations that occur as part of the unit process; and
- where the unit process ends, in terms of the destination of the intermediate or final products.

In clause 5.3.5 ISO 14041 (1998E) provides more detailed information on possible cut-off rules (see text box):

> During the scope definition, the initial set of inputs and outputs is selected for the inventory. This process recognizes that it is often not practical to model every input and output into the product system. It is an iterative process to identify the inputs and outputs which should be traced to the environment, i.e. to identify which unit processes producing the inputs or which unit processes receiving the outputs should be included in the product system under study. The initial identification is typically made using available data, and inputs and outputs should be more fully identified after additional data are collected during the course of the study, and then subjected to a sensitivity analysis (see 6.4.5).
> The criteria and the assumptions on which they are established shall be clearly described. The potential effect of the criteria selected on the outcome of the study shall also be assessed and described in the final report.
> For material inputs, the analysis begins with an initial selection of inputs to be studied. This selection should be based on an identification of the inputs associated with each of the unit processes to be modeled. This effort may be undertaken with data collected from specific sites or from published sources. The goal is to identify the significant inputs associated with each of the unit processes.
> Several criteria are used in LCA practice to decide which inputs to be studied, including a) mass, b) energy and c) environmental relevance. Making the initial identification of inputs based on mass contribution alone may result in important inputs being omitted from the study. Accordingly, energy and environmental relevance should also be used as criteria in this process:
> a) mass: an appropriate decision, when using mass as a criterion, would require the inclusion in the study of all inputs that cumulatively contribute more than a defined percentage to the mass input of the product system being modeled;
> b) energy: similarly, an appropriate decision, when using energy as a criterion, would require the inclusion in the study those inputs that cumulatively contribute more than a defined percentage of the product system's energy inputs;
> c) environmental relevance: decision on environmental relevance criteria should be made to include inputs that contribute more than an additional defined percentage to the estimated quantity of each individual data category of the product system. For example, if sulfur oxides were selected as a data category, a criterion could be established to include any inputs that contribute more than a predefined percentage to the total sulfur oxide emissions for the product system.
> These criteria can also be used to identify which outputs should be traced to the environment, i.e. by including final waste treatment processes.
> Where the study is intended to support a comparative assertion made to the public, the final sensitivity analysis of the inputs and outputs data shall include the mass, energy and environmental relevance criteria, as outlined in this subclause. All of the selected inputs identified by this process should be modeled as elementary flows.
>
> Source: ISO 14041 (1998E); clause 5.3.5.

Finally, clause 6.4.5 of ISO 14041 states (see textbox):

> Reflecting the iterative nature of LCA, decisions regarding the data to be included shall be based on a sensitivity analysis to determine their significance, thereby verifying the initial analysis outlined in 5.3.5. The initial product system boundaries shall be revised as appropriate in accordance with the cut-off criteria established in the scope definition. The sensitivity analysis may result in:
> – exclusion of life cycle stages or unit processes when lack of significance can be shown by the sensitivity analysis;
> – exclusion of inputs and outputs which lack significance to the results of study;
> – inclusion of new unit processes, inputs and outputs that are shown to be significant in the sensitivity analysis.
>
> Source: ISO 14041 (1998E); clause 6.4.5.

All these requirements are designed "to limit subsequent data handling". Their practicability is limited, however, for the data to be considered for cut-off must first be collected. Further data handling subsequent to data collection rarely poses a problem, in contrast to data collection itself.

From the ISO requirements it can be concluded that the cut-off problem is primarily a problem of data availability, with data collection consequently involving disproportionate use of resources. In all cases where data are available, no cut-off should be made. If data are lacking, there is a cut-off problem and this should be duly addressed.

In today's LCA studies capital goods are often cut off. Although capital goods can in principle be cut off like any other inputs (or outputs), there is a definition problem, in that agreement is lacking on what exactly constitutes a capital good, under what circumstances. In an LCA on a utility building the building will be the primary focus of the study, while in other studies it will be regarded as a capital good. It would appear more consistent, however, to treat capital goods the same as any other input or output flow. This approach has the key implication that industrial plant and equipment, as capital goods should, in principle, be included within the bounds of every LCA study (i.e. the processes required to build, maintain and decommission them). We therefore recommend that the same rules be applied for the cut-off of capital goods as for any other input or output flow.

In the past decade there have been a number of developments with regard to the issue of cut-off, beyond the terms of ISO. These can be divided into three categories:
1. avoiding cut-off by estimating flows using environmentally extended input-output modeling;
2. avoiding cut-off by estimating flows from similar flows for which data are known;
3. applying cut-off on the basis of predefined criteria.

Methods will be discussed below for each category.

1. Avoiding cut-off by estimating flows using environmentally extended input-output modeling

Economic input/output analysis (IOA) was developed by Leontief over fifty years ago. IOA proceeds from a so-called transaction table recording monetary flows between individual sectors of industry and the values added within each. In particular, such tables are used to represent the annual sales of each sector (to all others), offset by that sector's overall procurements (from all others). Transaction tables are converted into a matrix of coefficients in which each element of the table is expressed as a proportion of the total monetary output of the sector in question. Each column of this matrix represents the unique input characteristics required to produce the output in question. If it is assumed that coefficients are independent of production volume, the total direct and indirect inputs required for supplying a given external demand can be calculated by solving a system of linear equations. Under this assumption of linearity, the inputs required to meet $1,000 of external demand will be ten times greater than for $ 100 of demand.

Economic input-output models are applied in a variety of fields, including (economic) impact assessment, imputation assessment and energy analysis. Such models can be extended to include environmental data such as emissions and resource use per unit monetary output, on a sector-by-sector basis. The environmental interventions associated with satisfying a particular external demand can then be calculated by multiplying the interventions per unit sectoral output by the sum total of the direct and indirect inputs required for that demand.

An environmental input-output database has been developed at Carnegie-Mellon University, using a 1992 US input-output table and 1996 Toxic Releases Inventory (TRI) data (Lave *et al*., 1995; Hendrickson *et al*., 1998). Both the data and method were available at a website. Unfortunately, several important sectors such as electricity generation, mining and solvent recycling were absent in the TRI data. Moreover, the online service is no longer available at the moment. We can only hope that this situation will be remedied soon and practitioners are encouraged to follow these developments closely.

Suh has compiled an environmentally extended input-output table based on the most recent data sources available, including a 1996 US input-output table and TRI 98: Missing Inventory Estimation Tool (MIET). The data file is stored as an MS Excel spreadsheet in which all the direct and indirect environmental interventions are calculated by entering the estimated value of the flow in question. These results can be employed directly for estimating cut-off flows by adding them to the inventory. Support can be obtained online through http://www.leidenuniv.nl/interfac/cml/lca2/index.html. Alternatively, the hybrid model presented by Suh & Huppes (2000a) can be utilised for in-depth simulation. This hybrid model can be used to simulate full interactions between selected processes and the broader national economy embodied in the input-output table.

Although IOA provides a method for estimating lacking data, it has its limitations. In particular, practitioners should be aware of the following major shortcomings (Suh & Huppes, 2000a; 2000b):

- Input-output models provide information in aggregated form only and it may therefore be difficult to identify appropriate sectors to represent the missing flows. An IOA-based inventory for 'aluminium can' will yield the same result as for 'tin can' or 'iron can', since both are classified under the same IO code, 390100: Metal cans. Matters will be even worse if the product in question is located under aggregated classifications such as 020503: Miscellaneous crops, 110900: Other construction, or 570300: Other electronic components.
- IOA is based on assumed proportionality within the coefficient matrix, implying that economies of scale are ignored. Two scaling effects are relevant, however: input intensity and emission intensity per unit output, both of which generally decrease with increasing plant size. IOA gives the same amount of resource use and emissions per unit output regardless of scale.
- The base year of the latest version of the US input-output table is 1996. Given the dynamic nature of the modern economy, the economic structure of the day may no longer match that charted several years ago. Besides innovations in actual production technology, rapid development of environmental control technology and regulation are therefore also not taken into account. Although the errors due to this time lag have been reduced somewhat by using the latest (1996) input-output table, for some sectors this might be insufficient.
- In input-output tables, capital goods such as buildings, plant and equipment are counted as net outputs rather than inputs. That is to say, if a chemicals company builds a new facility or acquires new plant, the environmental interventions caused by those capital goods are not included in the total direct and indirect environmental emissions per unit monetary output of the overall chemical industry. The results given by an input-output based inventory model will therefore not include the environmental burden associated with capital goods as inputs.
- Potential errors may also be introduced through representation of data in monetary terms. Any analysis using the monetary input-output table assumes that the monetary flows in the input-output table precisely represent the actual physical flows between industries. This assumption implies perfect price homogeneity, which is not the case in practice. Secondly, monetary values must be converted into physical units for the purpose of LCA, which requires accurate valuation considering such matters as inflation.

In addition to these shortcomings, it should be noted that environmental IOA obviously cannot provide estimates where inventory data on the consumer use and/or post-consumer phase are lacking.

2. Avoiding cut-off by estimating flows from similar flows for which data are known

A second option for estimating the environmental effects associated with flows for which such data are lacking is to look at similar flows for which process data are available and determine the closest 'look-alikes' with respect to chemical structure, chemical properties, physical properties or other comparable properties. For example, missing data on an (in)organic catalyst used in a refinery process might be estimated from the process 'production of (in)organic chemicals' in the ETH database (Frischknecht *et al*., 1993/1995/1996). Capital goods in general might be estimated by subtracting the electricity production data of the SAEFL database excluding capital goods (SAEFL, 1998), from the same electricity production data of the ETH database but including capital goods (Frischknecht *et al*., 1993/1995/1996). Another method for estimating the impacts[1] arising from capital goods, particularly buildings, has been developed by Lindeijer (1998). In this method a rouigh estimate is made of the potential significance of the associated environmental impacts, using just a few data such as surface area, height, annual output and building lifetime. In this way an environmental impact profile is estimated by multiplying these figures by the individual profiles of reference construction parts, as calculated by the Eco-Quantum software.

These kind of estimates can obviously be performed at various levels of sophistication. For example, a practitioner could consult an expert, a chemist say, on which flow is the most reasonable 'look-alike' to the flow for which the data are lacking with respect to chemical structure. The capital goods method developed by Lindeijer (1998) has also been developed at three different levels of sophistication.

As there is no documented method for most of the options discussed above, it is difficult to discuss this issue in general terms. In general, non-documented options are likely to involve considerable arbitrariness, as expert opinions on what is 'reasonably similar' may differ significantly from case to case and practitioner to practitioner.

[1] Note that these estimates would be given as environmental profiles, in terms of indicator results for abiotic depletion, global warming, ozone depletion, etc., and not as inventory results.

In addition, the capital goods method described provides only a rough indication of actual environmental impact. The uncertainties associated with the various input data used in this approach have been assessed; for further details, see Lindeijer (1998) and Part 2b, Section 3.7 of this Guide.
Another important practical drawback of this method is that it estimates environmental impact profiles using the Impact assessment methods proposed by Heijungs *et al*. (1992). As new Impact assessment methods are recommended in the present Guide, this implies that the capital goods method would yield results that are incompatible with the Impact assessment results of the remainder of the LCA in question. This method can therefore not be recommended here.

3. Cut-off based on predefined criteria
Cut-off criteria like those mentioned in ISO 14041 (1998E) are in general use by LCA practitioners today to decide which flows will be studied and which to exclude. In the Nordic Guidelines (Lindfors *et al*., 1995a,b) three different principles are mentioned for introducing upstream cut-offs (i.e. those not involving waste management processes):
- cut-off at a predefined upstream stage of the lifecyle;
- cut-off at a predefined mass percentage of the input flows associated with each individual process;
- a slightly modified version of the second principle, with cut-off of upstream input flows only when they fall below a certain, predefined percentage of the total mass inputs of the product system (see ISO/TR 14049, 1998).

The authors mention that "these cut-off criteria should only be applied for common emissions for which associated environmental impacts mainly depend on mass flow and not on quality. As a default approach this type of cut-off criteria may only be used for energy related emissions." For certain emissions a flagging procedure is proposed.

In a report on application of LCA to agricultural products (Wegener Sleeswijk *et al*., 1996) discussion of the cut-off issue focused on capital goods. In the agricultural sector a variety of capital goods are employed with a relatively short service life (e.g. farming machinery) as well as capital goods requiring relatively large quantities of materials (e.g. farm tracks and roads). The authors argue that the environmental interventions associated with machinery production and maintenance as well as with farm tracks and roads should not therefore be omitted from LCAs on agricultural products. Farm buildings, on the other hand, can generally be excluded, except in the case of greenhouse horticulture and in studies where farm buildings constitute the main issue.

PROSPECTS
In a PhD project CML (See Suh & Huppes, 2000a) is working on a hybrid input-output model, which may be useful for estimating missing data in the LCA context. This hybrid model can be used to simulate full interactions between selected processes and the broader economy. However, the limitations of preparing inventories on the basis of input-output analysis should be clearly recognised by users and this type of estimate should be kept to a minimum, particularly if estimated flows prove to be significant. Developments on this issue will therefore need to be updated in this Guide in due course.

CONCLUSIONS
If data on specific process flows are lacking, arbitrary cut-offs should be avoided wherever possible by using suitable methods of approximation:
- This allows an initial estimate to be made, permitting a more reasoned decision on whether to collect process-specific data for the flows in question or subsequently ignore them.
- Environmentally extended Input-Output Analysis (IOA) is generally recommended as an approximation method, because of its broad applicability (almost all flows) and completeness in terms of system approach.
- If comparison to a similar process (based on expert judgement, for example) is anticipated to provide a better estimate than IOA or if IOA is not applicable (use and waste management phases), this may be used alongside or instead of IOA.

RESEARCH RECOMMENDATIONS
- To avoid mere cut-off with respect to the boundary between processes to be included and those to be ignored, appropriate procedures need to be developed based on estimates of the environmental interventions associated with these processes. Input-output analysis is one of the most promising avenues of research here. Since the Carnegie Mellon and the Suh and Huppes (2000a; 2000b)

models are based solely on US input-output data, there is a need to develop a more general (average-world) model or at least regional (e.g. European and Asian) submodels.

3.9 Multifunctionality and allocation

TOPIC

Most industrial processes are multifunctional. Their output generally comprises more than a single product, and raw material inputs often include intermediates or discarded products. In other ways, too, production processes are dynamically interlinked with other processes, technologically, behaviourally, and otherwise. LCA practitioners are thus faced with the problem that the product system or systems under study provide more functions than the one investigated in the functional unit of interest. An appropriate decision must therefore be made as to which of the economic flows and environmental interventions associated with the product system under study are to be allocated to (the functional unit provided by) that system. Decisions on the specifics of allocation will obviously be determined by the precise nature of the system boundaries as previously defined (see Section 3.2), for these determine which inputs and outputs are to be taken as being associated with the function of interest. An appropriate allocation procedure is thus required to partition the inputs and outputs of all relevant processes to the appropriate product systems.

In comparative LCAs, the problem is compounded, for any comparison of alternative product systems for fulfilling the function of interest is hampered by the fact that each of these alternatives will be associated with a variety of different additional functions, in addition to the function of interest. As an example, cadmium production from zinc ore yields zinc as a co-product, while production from phosphate ore yields phosphorus as a co-product. The aim of the allocation step is then to render the two production systems 'equivalent', for which two basic methods are available. Equivalence can be achieved either by subtracting those parts of the systems that function for other product systems, resulting in a system providing a single function (viz. the primary function), or by adding further subsystems such that the two systems provide the same set of functions.

Viewed from this angle allocation is essentially a process-level problem, closely related to the definition of system boundaries. At the same time the issue of multifunctionality is far broader, however, having a bearing on the entire issue of inventory modeling at the core of LCA. In attempting to map the complexities of the production, consumption and waste management systems embedded in the dynamic real-world economy, the models employed in LCA inevitably introduce a host of simplifications. At present the chief goal is still to model basic process-technological interlinkage, using fixed input-output coefficients and steady-state inventory models and provisionally ignoring any further relations of a social, cultural and political nature. Although progress is being made on incorporating some economic mechanisms, the issue is a complex one that raises basic questions about modeling choices. In this sense, then, process multifunctionality can be regarded as a modeling problem at the systems level. In more technical terms, and assuming purely linear relationships,, there is then an imbalance between the number of equations and the number of variables in the model, to be solved by adding or subtracting equations or variables.

There are thus two aspects to what is loosely referred to as allocation: an ‘allocation problem’, in the narrow sense of partitioning (the inputs and outputs of) unit processes among product systems, and a ‘multifunctionality problem’, a broader issue cutting right across LCA inventory modeling. These problem definitions are aligned with basic choices about where physical causality is to be incorporated in the LCA procedure: during modeling, or at the allocation stage. As inventory models are further refined to more broadly mirror real-world causality and hence become more complex, the problem of multifunctionality will be compounded. Then, too, Some form of allocation will therefore remain unavoidable and an appropriate, pragmatic procedure will have to be adopted. Given the additional fact that much of the work on allocation is closely allied with developments in LCA modeling, the topics of allocation and multifunctionality are treated together in this section.

Below we first summarise relevant sections of ISO Standard 14041 (ISO, 1998E), which provides a stepwise allocation procedure for practitioners. Next we turn to the 1992 guide (Heijungs *et al*., 1992),

which is oriented more towards multifunctionality as a systems (i.e. modeling) problem. In these introductory paragraphs we also outline the basic contours of the problem in its dual perspective and consider some ambiguities of terminology. There follows an analytical and strategic review of recent developments in LCA modeling and allocation. This serves as a stepping stone to a statement of the principal choices made in this Guide, which are then related to the ISO allocation framework. The recommended procedures are described in greater detail in the 'conclusions' of this section, where some of the main options available for sensitivity analysis are also indicated. The topic of multifunctionality and allocation is also treated separately in Appendix C.

DEVELOPMENTS IN THE LAST DECADE

ISO

The topic of allocation is treated in ISO 14041(ISO, 1998E). In clause 6.5.1 the basic terms of the issue are set out as follows:

"Life cycle inventory analysis relies on being able to link unit processes within a product system by simple material and energy flows. In practice, few industrial processes yield a single output or are based on a linearity of raw materials inputs and outputs. In fact, most industrial processes yield more than one product, and they recycle intermediate or discarded products as raw materials. Therefore, the materials and energy flows as well as associated environmental releases shall be allocated to the different products according to clearly stated procedures."

In the terms of the present Guide: in the case of multifunctional processes, the associated environmental interventions are to be appropriately allocated to the product systems under study.

In Section 6.5.2 of ISO 14041 several principles for allocation are then stated. In the first place, allocation procedures are to approximate fundamental input-output relationships as closely as possible. Second, the sums of the inputs or outputs of a multifunctional unit process allocated to its various goods and services are to equal the unallocated inputs or outputs of that process. This is known as the '100% rule'. Where several alternative options for allocation are available, finally, a sensitivity analysis is to be conducted to illustrate the consequences of the particular choices made.

Section 6.5.3 lays down an allocation procedure comprising three steps. Step 1 does not, strictly speaking, belong to the allocation procedure, as its aim is to avoid allocation "wherever possible", by either of two options: division of multifunctional processes into two or more monofunctional subprocesses (step 1a), or expansion of the product system to include the additional functions related to the co-products (step 1b). The latter step essentially means redefining the functional unit and the system boundaries, and must therefore be conducted in accordance with the provisions for defining these basic parameters. Where allocation is unavoidable, ISO prescribes one of two alternatives. Step 2 states that system inputs and outputs should be "partitioned on the basis of the underlying physical relationships between them", i.e. reflect the way in which the inputs and outputs are affected by quantitative changes in the products or functions delivered by the system If this is not feasible or does not provide a full solution, step 3 of ISO 14041is to be followed, with partitioning being based on "other relationships" between inputs and outputs, most notably relationships of economic value.

Avoidance of allocation, the first step of the ISO procedure, is quite simple in variant 1a, as when truly independent processes are lumped together into one unit process. Specifying these underlying single processes then solves the problem. In variant 1b, system expansion, the solution is more problematical than it may seem at first glance. Consider the case of a comparison between product system alternatives (as opposed to a single alternative LCA). To create 'equivalence' between the functions provided by these alternatives, in principle one may simply add to each alternative all the additional functions pertinent to the other alternatives. With a large number of alternatives, and with more detailed analysis of underlying multifunctional processes, however, such 'system expansion' will lead to a new de facto functional unit comprising a vast number of extraneous functions. The system as a whole may thus become inordinately large and be quantitatively dominated by all the added processes, with all their attendant uncertainties. One partial solution to this problem is not to add functions to the various alternatives but to subtract them from those alternatives providing additional functions. This approach to ISO's system expansion is also known as the 'substitution method' or 'avoided burden method' and is discussed in greater detail below. Thus, in a comparison of, say, alternatives for soap production from caustic soda, the upstream monofunctional chlorine production chain is not added to all the alternative

product systems, but is subtracted from the former. From an economic perspective, however, this procedure can be regarded as product substitution, the additional chlorine from soap production replacing monofunctional chlorine production elsewhere. It thus involves a number of implicit assumptions on which material production is replaced, on the absence of new market demand arising through substitution (i.e. only substitution within existing markets) and on the feasibility of other multiple outputs within the system being able to be dealt with in the same way. If these assumptions are unfounded, i.e. if such substitution is unlikely to occur in reality, subtraction becomes an artificial procedure, adopted solely to yield a monofunctional system. This is indeed the case in the soap example, for in reality there is no such thing as a monofunctional chlorine process. Thus, there is considerable ambiguity between the terms of ISO's step 1b: system expansion and step 3: allocation based on "other relationships", viz. economic. These issues will be considered in more detail below.

The requirements set by ISO on this kind of subtraction procedure are that the alternative systems to be subtracted should be “known” and that assumptions about what is actually replaced by its output be “well-documented” (ISO 14041, 1998E; B.2). As we have seen, though, this is often problematical, for with many joint products there is no independent production process to add or subtract. Moreover, if the substitution system is itself multifunctional, the multifunctionality problem remains, in the line of reasoning of ISO step 1b, again to be tackled by subtraction, for each of the multiple flows. Even if applied only partially, in cases where realistic alternatives are indeed known, system expansion would lead to endless regress, involving virtually all the world’s production processes, for the simple reason that virtually any substitution system is itself a complex multifunction system.

It should be reiterated here that we shall interpret the ISO procedure within the dual framework discussed earlier: in the context of the empirical modeling of all kinds of process multifunctionality, and as a solution to the multifunctionality problem, which by definition then is not empirical modeling, although it may of course reflect empirical relationships in a looser sense. Most ISO steps may be interpreted either as modeling or as partitioning, as a solution to the multifunctionality problem.

Division of multifunctional processes (ISO step 1a) means taking a closer look at empirical relationships and is clearly part of modeling. Given the implications for redefinition of the functional unit and system boundaries, system expansion (step 1b) is also treated by ISO implicitly as a modeling step. It represents the acceptance of multifunctionality. If viewed as a solution to the multifunctionality problem, expansion of the functional unit to include the functions of the co-products means that these latter functions must be incorporated as separate production chains in all the product alternatives that are not associated with these co-products. Particularly if there are several alternatives, each with several but different co-products, the perspective on the original functional unit soon becomes clouded as the system is expanded to include ever more functions. The situation is illustrated in Table 3.9.1. System expansion renders systems comparable by adding to the respective product systems co-products that are deemed 'equivalent' (A' for A, etc.). This yields a series of hypothetical systems providing the same, multiple functions. Since subtracting a constant factor from (the functions associated with) each alternative leaves the differences between the alternatives mathematically unaltered, one way of reducing complexity is to simply deduct from each alternative the sum total of all the ‘extra’ functions (the set A’+ B’ + C‘), yielding product systems assumed to be monofunctional, with just a single co-product (system) subtracted.

Table 3.9.1: Expansion and subtraction as solutions to the multi-functionality problem.

Intended FU (X)	co-product	expanded system : multiple but equivalent functions	subtracted system : one function only
alternative system X_1	A	X_1 + B’ + C‘	X_1 – A’
alternative system X_2	B	X_2 + A’ + C’	X_2 – B’
alternative system X_3	C	X_3 + B’ + C’	X_3 – C’

As already mentioned, the problem with this kind of subtraction is that it can be interpreted not as a solution to the multifunctionality problem but as a step in modeling. It is then not ‘subtraction’ but ‘substitution’, i.e. a modeling of the economic substitution of processes. The implicit logic is that if electrical power is generated as a co-product in some waste management process this will lead to reduced power generation elsewhere. There is unjustified optimism about (system) expansion and

(function) subtraction as a solution to the allocation problem. There are two main drawbacks. First, system expansion generally means adding new multi-functional processes, thus merely increasing the number of processes that need to be partitioned. Second, this procedure may often constitute an artificial solution to the multifunctionality problem, if the functions taken for expansion and subtraction are known in reality not to be the relevant ones. This is not a very elegant solution, as such imaginary solutions may introduce large and unknown uncertainties in outcomes. A practical solution is to employ system expansion as a proxy type of modeling, expanding the system using processes that are not precisely equivalent but are similar enough for them to be taken as reasonable approximations. These processes should already have been rendered monofunctional by means of some allocation procedure, in a prior study, for example. For simplified LCA studies this may indeed represent a useful mixed solution: system expansion on the basis of previous allocation. A frequently employed option for subtraction is to use the allocated cradle-to-gate (partial) LCAs in the ETH energy production database (Frischknecht *et al*., 1993/1995/1996).

Allocation based on physical relationships (step 2 of the ISO procedure) is part of modeling if these relationships are indeed specified, as when a waste management model specifies emissions as a function of the input of some waste. This is a clear example of modeling in which the multifunctionality problem does not arise. (Whether this type of modeling is adequate for all or even most LCA purposes is another question.) If such empirical relationships are not modeled, some physical relationship may still be used for partitioning, as a somewhat artificial solution. The same holds for the final ISO step, partitioning on the basis of "other relationships", such as economic value. Market mechanisms may be modeled, and are then part of the modeling leading to the multifunctionality problem, or they may be ignored.

The point here is that if there is relevant real-world knowledge available on empirical relationships, it should be used not only for allocating multifunctional processes, but also, at a primary level, for constructing a more accurate and consistent inventory model. ‘System expansion’, in the real world, is part of a broader class of economic substitution mechanisms, involving market mechanisms, income effects, investment decisions, etc. As economic mechanisms are clearly fundamental to industrial processes, model quality would be greatly improved if market mechanisms were somehow incorporated. At the same time, though, introducing economic mechanisms would increase the complexity of LCA inventory modeling enormously and compound the problem of multifunctionality still further. It may even be questioned whether such modeling can be reconciled with certain key features of LCA, such as an arbitrary amount of the functional unit.

The steps of the ISO allocation procedure and their interpretations in this Guide are summarised in .

Table 3.9.2: ISO terminology and its interpretation in this Guide.

ISO terminology:	interpretation in this Guide:
allocation problem: at process level	multifunctionality problem: at unit process level or at systems level
solution	
Step 1: avoiding allocation:	
1a division of unit processes	modeling
1b system expansion	modeling of broader functional unit, or: modeling of economic substitution or: non-empirical, artificial solution
Step 2: otherwise, allocation reflecting physical relationships	modeling, or: allocation as partitioning of unit processes
Step 3: otherwise, allocation reflecting other relationships	modeling, or: allocation as partitioning of unit processes

discussed. Although the problem was defined at a systems level, solutions were discussed at the process level, in conformity with ISO. System expansion was not deemed a satisfactory solution, for incorporating ever more processes, themselves multifunctional, leads to highly unwieldy systems that are extremely difficult to compare. Individual processes were described in terms of inflows and outflows, some of which were to be allocated and others allocated to. Although "social causality" was stated to be the guiding principle, it is not always applicable, for practical or theoretical reasons. Social causality refers to the aims of operators, who adjust operations to shifts in demand and more generally to supply and demand mechanisms, influencing the product mix of multifunctional processes. The 1992 guide defines the multifunctionality problem in relation to system boundary definition. Co-products not intrinsic to the system under study need not be followed further: they cross the system boundary. However, if outputs are produced in a waste management system that can usefully be applied in other systems, as with some kinds of fly ash and sewage sludge, there is no multifunctionality problem; the flows should then be followed further in the processes where they are applied. Open-loop recycling is treated at the systems level: primary production is assigned to the first system of use, with all upgrading processes being considered part of the second system of use. Final waste processing, including landfill, is assigned to the process where it actually occurs, which is generally multifunctional. Although allocation rules for multiple waste processing were discussed in the 1992 background document, no clear procedural guidelines were given in the guide.

Compared to the 1992 guide, the current Guide has more precise rules for setting systems boundaries, in relation to the economic value of flows. It is also more explicit with respect to the allocation rules to be applied, as elaborated below. Moreover, the rules for solving the multifunctionality problem are now more uniform across the three situations distinguished in 1992: co-production, combined waste processing and open-loop recycling.

Other methods

A general survey of developments in the field of allocation is provided by Lindeijer and Huppes (see Appendix C) and specifically for open-loop recycling by Ekvall & Tillman (1997), Kim *et al*. (1997) and Klöpffer (1996). We shall not discuss all these developments individually but review the field from a more strategic and analytical perspective, for unfortunately we are here not dealing with refinements of existing positions, paving the way to ever better, more detailed solutions. In Kuhnian terms: there is still no dominant ‘normal science’ paradigm in LCA, but several competing paradigms, all rather imprecisely defined. These paradigms are not reflected or defined solely at the level of allocation. As already indicated, the multifunctionality problem is intimately related to the more fundamental issue of inventory modeling. The following analysis of developments in the last decade therefore covers both allocation-as-partitioning and inventory modeling. As the various approaches to the problem involve paradigmatic choices, they have one feature in common: they tend to lead to improvements in one area while creating new problems in others. A second issue is that causal relationships which have been incorporated in modeling cannot be used once more in allocation. Consequently, choosing a method for tackling the multifunctionality and allocation problem is not merely a question of single-step improvements.

Analytic survey: trends

This review concerns allocation and modeling. The more general subject of modeling for LCA and, still more general, for Industrial Ecology, is touched upon only in relation to the multifunctionality problem as specific to LCA. Certain broader topics relating to LCA modeling have been treated in Section 1.2.2 above.

LCA inventory analysis has progressed in two main ways over the past decade: more specific methods and models have been developed, and there are now clearer ideas on how to interpret and compare their results. The basic aim of all this work has been to include more causal mechanisms of known relevance in inventory models. At the same time, though, there is a general conviction that LCA should be simplified, enabling easier and hence broader application. Up to a point, the conflicting aims of making LCA both ‘better’ and ‘simpler’ can be practically resolved by using better databases and software. These may reflect enhanced methods, on the one hand, and are easy to apply, on the other. At the methods level the conflict remains, however. For this reason, in this Guide a distinction has been introduced between ‘simplified’ and ‘detailed’ LCA. Below, we first consider detailed LCA, which employs more sophisticated modeling and allocation methods. After this fairly lengthy treatment, a simplified form of analysis is derived that still retains as much realism as possible.

Certain simple types of modeling may even avoid the problem entirely, as when economic processes are characterized in monetary terms only, using sectoral input-output models with environmental extensions, for example (Lave *et al*., 1995; Hendrickson *et al*., 1998). As a gross simplification, this type

of modeling may be used in LCA, thus rendering the multifunctionality problem invisible. We regard the application of such IO models as useful for estimating missing data but not as a general modeling approach (see Section 3.8), nor as a solution to the allocation problem.
It is sometimes proposed to include exergy analysis in LCA inventory modeling and solve the allocation problem by means of this single measure on product flows (and even environmental flows), as seems to be advocated by Ayres (1998) and, to a lesser extent, by Cornelissen (1997). Exergy analysis may constitute a useful tool for identifying strategies for improving energy efficiency, as improved system alternatives. If such a strategy has then indeed been effective in reducing various forms of energy resource use (e.g. oil, coal, uranium) is a matter of LCA inventory modeling of the alternatives concerned, comparing these, and other, environmental interventions. In the present discussion of multifunctionality and allocation we give no further consideration to the issue of exergy analysis.
Other types of model may aggravate the allocation problem substantially, as is the case with most partial-equilibrium economic models. Substitution, for example, is in reality always only partial. Shifts in volumes and prices in one process will thus lead to adjustments in the volumes and prices of all related processes, and so on. If processes are defined at the level of detail required for environmental analysis, the 'full system' can never be modeled. Increased modeling complexity is therefore combined with incompleteness, apart from compounding the allocation problem. One solution to the multifunctionality problem is to simply accept the fact that systems are multifunctional and thus avoid the additional artificiality of allocation. This solution comes at a price, however. It will then be harder to compare product system alternatives, as differences in a multitude of other functions will have to allowed for when interpreting the modeling results. This amounts to abandoning the functional unit: the very essence of what constitutes an LCA.
Most current LCA inventory modeling relies on three basic simplifications, all more or less directly related to ease of use. First, unit processes are treated as ‘black boxes’, with constant and linear input-output coefficients, in which the actual variability of process parameters and operating conditions are thus implicitly disregarded. A second, related simplification is that no attempt is made to incorporate the actual objectives of process operators, i.e. plant owners, managers and investors. As a consequence, changes in exogenous circumstances do not lead to endogenous changes in process parameters. Third, and again related to the previous simplifications, the inter-connectedness of processes is also modeled very mechanically and statically, in a steady-state model, ignoring all (or most) market mechanisms, as well as all other social, cultural, and political relations.

These basic simplifications are obvious limitations, and overcoming them is a worthy endeavour, though by no means a straightforward one. As most recent efforts to elaborate a more sophisticated allocation method have been closely bound up with the goal of improving LCA inventory modeling, recent developments can usefully be discussed in terms of efforts to tackle the respective simplifying assumptions. We shall therefore consider, successively, efforts aimed at incorporating the following three classes of mechanisms in inventory modeling:
1. technical relations;
2. aims of process operators;
3. market relations.

Technical relations
Today there is a vast body of technological knowledge that can be usefully incorporated in LCA. Systems engineering models of complex systems have become increasingly accurate and a variety of sophisticated databases and software are now available, very similar to LCA software, such as Chemsys (Chemstations Inc., 1997). Generally, they comprise a mixture of physical ('natural science type') causality with operational practice based on technical and economic aims. In the field of waste management, process models with ‘internal relations’ have been developed to determine the effects of processing an extra amount of a particular waste flow; for a review, see Sundberg *et al*. (1998). A number of technological models have also been developed for specific use in LCA; see, for example, Eggels & van der Ven (1995). Such models have also been incorporated in LCA software. In models permitting independent variation of useful functions, these technical specifications may help solve the multifunctionality problem or, better, avoid it altogether. A change in one of the outputs associated with a functional unit can then be related to changes in the overall process, leaving all other functional outputs constant. It is then these changes that are due to the functional unit. If the co-products cannot be varied independently, i.e. if it is a joint (rather than just combined) process, the multifunctionality problem of course remains.

There is one major problem with recent developments along these lines. Most available models indicate the results of short-term changes only, computing effects on capacity utilisation while leaving installed capacity unchanged. While such models may be very useful in optimising use of existing installations, most of the questions posed in LCA relate to the medium and long term, in which, conversely, installed capacities change, with capacity use more or less given. This partial 'solution' to the allocation problem therefore comes at the price of an unrealistic model. The obvious but by no means straightforward solution is to model long-term technical relations, incorporating changes in installed capacities and in operational utilisation due to particular choices of product system. Both the short- and the long-term analysis are forms of marginal systems analysis; in the short-term analysis it is marginal changes in capacity use that are relevant, with installed capacities remaining constant; in the long-term, it is marginal changes in installed capacities, at intended capacity use, that are relevant. So long term marginal analysis specifies the changes in average functioning. See also Section 1.2.3.4.

There is wide debate on how best to model causality within process relationships, treated in the context of allocation in ISO 14041 only partially under the heading 'physical relationships'. These relationships define the input and output coefficients of all the processes included in the LCA inventory. However, these 'technical' coefficients also reflect the aims of process operators, and these are mainly economic. Furthermore, 'physical causality' is generally in the wrong direction when applied in LCA: in a physically determined system it is the inputs and system conditions that cause the output, while LCA is concerned with how the multiple products delivered by a given process, as outputs, affect its functioning, including inputs of raw materials and intermediate products. From this perspective, the actual 'physical relationships' embodied in the technological plant are generally of no more than secondary influence. There may be one exception here: where the 'cause' coincides with the 'function', as in waste processing, where the wastes imported are the precise cause of the emissions and other outflows of the plant in question. Even in this case, though, the inputs of capital and ancillary goods are not physically determined but induced by regulations and a variety of socio-economic factors. Pure physical causality can never fully explain the functioning of even waste management systems. In mixed waste treatment, in particular, the rationale for co-processing often lies in economies of scale, with consequent acceptance of inferior (because unspecialised) treatment. From this broader perspective, there is good reason to use economic allocation, i.e. the share in the total proceeds of the waste management process represented by sum paid for processing this specific waste. As already discussed, there are thus two conflicting options here: to incorporate physical, i.e. technical causality in modeling, thereby possibly resolving the multifunctionality problem, or to resolve the problem as part of an allocation procedure. The solution proposed by Eggels & van der Ven (1995) is a mixed one. It distinguishes between product-related and process-related flows and emissions. Product-related emissions indicate what would happen if the specific waste flow were not part of the total waste flow (incremental change, with fixed installed capacity) or if a unit more or less were processed (marginal change, also with fixed installed capacity). This modeling is generally done on theoretical grounds, e.g. explaining cadmium emissions to air in terms of the cadmium content of the waste products being processed. By relating all emissions to their sources in specific waste products, only part of total emissions are explained, termed by them 'product-related emissions'. Stopping here would mean violation of the 100% rule, i.e. total allocated emissions would not equal total emissions. In a following step, then, the remaining unallocated emissions, termed 'process-related emissions', are allocated on a different basis: by mass, for example. The situation becomes slightly more complex if, say, electricity is generated alongside the waste processing function. By first substituting a stand-alone electrical power plant for the co-produced energy, the same procedure can be applied. First the product-related emissions are established and then the process-related emissions, which may now be negative, and both allocated to the various waste flows being processed.

One drawback of this method is that in most cases the physical-causal analysis used to establish the product-related emissions relates to the short term, with fixed capacity and variable capacity utilisation, while LCA is concerned primarily with the longer term, with fixed capacity utilization and variable installed capacity. Even if one were to accept the validity of a short-term analysis as a proxy for the long term, the ins and outs of that analysis remains unclear. How to tackle the PVC contribution to dioxins emissions from a waste incinerator in which other chlorine-containing wastes (e.g. kitchen wastes) are also being processed? Current models indicate that the marginal contribution is negligible, as the chlorine input required to produce all the dioxins is much lower than the actual input, from either PVC or

kitchen wastes. The marginal contribution to dioxin formation, the basic method for establishing physical causality, is therefore virtually zero.
Clearly, more work needs to be done before process models properly reflect the underlying internal technical relations. The task then is to elucidate both the modeling principles required to handle long-term capacity adjustment and the precise empirical relationships that need to be incorporated in the model. For the time being, realistic modeling of physical relationships within the process is an option in exceptional cases only, relating to waste management. For establishing long-term relationships involving changes in installed capacities, no clear approaches to improved modeling as yet exist. Thus, the multifunctionality problem must be resolved mainly as part of the allocation procedure, rather than on the basis of physical relations.

Aims of process operators
Most industrial processes have many degrees of operational freedom, over the shorter term but particularly in the long term, as investments in new capacity may fundamentally change the nature and scale of operations. To incorporate the aims of process operators in LCA therefore appears to be a promising methodological extension. There is indeed a wealth of knowledge on the aims of business enterprises, which might be expressed in terms of a mixture of goals as e.g. market share, cash flow, profits, shareholder value and long-term viability, as defined by the firm in question. Combined in a goal function, an optimum can be defined for the functioning of the process (or set of processes) operated by a firm, by maximising its goal function.

It is process operators, usually the owners or their representatives, who decide how processes are operated. Their aims are primarily economic, viz. to minimise cost or maximise net proceeds or profits. Such types of model are sensible only if used in tandem with models having specified internal technical relations. Assuming some goal function which operationally specifies the aims, an optimum system can be developed. The first assumption here is that there is one operator for the entire system, the second that the relations specifying the system include options for capacity adjustment, to allow for the long-term perspective relevant in LCA. Then, during optimisation, capacity use is set at the desired optimum in all installations. Adding one functional unit gives the long-term marginal effect required in LCA. For all combined processes, but not for joint processes, the multifunctionality problem is then, in principle, solved. However, there are few situations in which there is just one operator maximising his goal function, and the vast majority of processes are at least partly joint in nature.

An example in a relatively simple situation is that of a mining industry. Azapagic (1996) and Azapagic & Clift (1998, 1999, 2000) internalise in the model the aims of process operators using the methods employed by economists in operations research and, more generally, in production function theory (Heijungs, 1998b). Given currently available technologies, an optimum input and output mix is chosen on the basis of the assumed goal function of the operators. Because short-term technological relationships are employed, there is real progress towards solving the multifunctionality problem, as the model specifies the effects of providing an additional amount of function. For the very same reason, however, all system responses are calculated within the constraints of installed capacities, while the question of interest in LCA usually is how the world would be changed by adjusting those capacities. There is little experience with this approach in the context of long-term studies, moreover. Unfortunately, then, this kind of optimisation model has only limited relevance for most kinds of LCA. As the variations due to extra unit production of one of the products now result from investments in real production plant through added installed capacity, the multifunctionality problem remains much the same. The short-term optimisation models now available represent an initial step towards realistic short-term modeling of dynamic markets. Having laid down technical, i.e. plant parameters and a goal function, supply functions can in principle be specified. Although a number of large companies dispose over such models, they comprise little technological detail (publicly at least). The optimisation models used for allocation assume that all decisions regarding process adjustments are made by a single operator, based on his own particular goal function. In reality, of course, product systems have a multitude of different operators, each with their own goal function, and a multi-actor model is required. As the choices of each actor are influenced by those of all the others, model complexity soon spirals as the number of actors grows. The goal function and the process conditions may cover the long term, as is generally required in LCA. As long-term models employ technological averages, however, they do little to help resolve the multifunctionality problem. First, there are vastly more aggregated processes requiring subsequent

allocation. Second, in a given situation any change in overall output will affect only a limited number of constituent processes (under the simplifying assumption of linear relations).

If this assumption of a single actor is abandoned, several new problems of a game-theoretical nature arise, for the choices of any one actor are influenced by those of all the others. Although economists are developing such multi-actor models, their level of technological specification is too general for the purposes of LCA. Technology-specific, multi-actor models become feasible if, e.g. an assumption of fixed prices is introduced. Then interdependency is very much reduced. If such simplified multi-actor systems are optimised economically, e.g. in terms of minimal costs, combined (but not joint) processes can be analysed as to optimal functioning. For each alternative for the functional unit, the model would indicate installed capacities and the optimum ratio between the various product outputs given this installed capacity. This means modeling large systems, as the co-products are then also modeled. In such models, the number of allocation problems might be reduced somewhat. For primarily joint processes there would still be no solution, as the ratio between product outputs is then more or less fixed. In the context of LCA there is no systematic work currently in progress in this direction.
When adjusting installed capacities, there are usually several technological options available; see, for example, the many options for electrical power production. Endogenising such choices based on optimisation rules is an endeavour that has not yet been undertaken in LCA. There is an ongoing debate on the more limited subject of the marginal process choice. Clearly, a process type that cannot be extended, as in many places hydropower, is not a relevant option for additional capacity. Only non-constrained processes are relevant, with state-of-the-art processes as a further focus. Such technology can be viewed as a proxy for the processes that will be functioning in the not too distant future. Processes that have been introduced on an experimental basis only, like chlorine-free polycarbonate production or fast-breeder nuclear power generation, are excluded, as are older technologies that are still operational but no longer the object of investment. A similar position is adopted by Weidema *et al*. (1999) in their notion of the 'marginal process', which likewise limits the set of potentially relevant processes to include only those that can with certainty be perpetuated in the longer term. Even though there are currently no limitations on installing hydroelectric power plant, for example, use of hydropower cannot be extended indefinitely because of the limited supply of water for this purpose. In both these approaches the choice of process is exogenous. Modeling which processes are current state-of-the-art is not yet possible, for. such information is currently only exogenously given. There is no particular reason to assume that state-of-the-art processes are single function processes. On the contrary, modern processes will tend to avoid waste and produce additional products. This brings modeling of the marginal process back to the position adopted in Heijungs *et al*., 1992, where the marginal process was defined as the current–state-of-the-art process (then named: modal modern process). Such a marginal process choice is not related to the aims of the process operators. No progress is made in resolving the allocation problem, however, for a marginal or current state-of-the-art process is as multifunctional as any other (if not more so, given the aim of industrial ecology to integrate waste flows into production processes.

One special case of optimisation at the systems level is where it is not linked to the goal function of the firm or firms involved, but to a social goal function such as a 'sustainable emission level' (Azapagic, 1996; Frischknecht, 1998; Frischknecht, 2000), involving social, economic and environmental goals. Two interpretations are then possible. One is that the answer given specifies the socially most attractive option. This answer does not indicate what will happen, or, more low key, in which direction developments will take place. It indicates a most reasonable environmental potential of some set of technological options. The second interpretation (as in Frischknecht) is that, in the longer run, public policy will be adequate and society will arrive near the optimum for any technology chosen. The model then has an empirical predictive value, taking into account politico-administrative mechanisms as yet unspecified. Again, as with optimisation based on the goal of the firm, long-term models of this kind may give some empirical insight but have little to contribute to resolving the multifunctionality problem.

Finally, there is the option not to expand model mechanisms with real-life goal functions of firms or society, but to include one, simplified goal of the firm, not in empirical modeling but in the interpretative allocation procedure (Huppes, 1993). As most goals of process operators relate to sales, the share of each product in total sales of the firm indicates its share in bringing about the existence of the full (unallocated) process. These shares in proceeds may be used as allocation factors, allocating all flows to each of the products/functions, proportionally to their share in overall monetary proceeds. This

‘economic’ allocation procedure does not indicate the partial effect of solely delivering some extra amount of the product under investigation; it indicates its share in overall effects. Modeling is thus concerned with establishing the overall multifunctional effect of a change in demand for the functional unit, and allocation with establishing the share of each product in that overall effect.

Market relations

Given the crucial role of market relations in the systems-level effects of the technological choices being analysed in LCA, use of a market model comprising inter-process market relations would represent a major step forward. There are a number of developments of interest, originating both within LCA and from the field of (ecological) economics, and these will be surveyed in turn. First, however, let us recapitulate on the extent to which market relations have already been incorporated in inventory modeling and how this relates to resolution of the allocation problem. The system expansion step in ISO can be interpreted as economic substitution, being the special case of full substitution. Economists are well aware that there is rarely full substitution, as implicitly assumed in the avoided burden method (as explained in the discussion of ISO 14041, above). Only if demand is extremely elastic will changes in supply not lead to market adjustments . And even if there is a "known alternative" available (as required by ISO), to assume full substitution is still unrealistic. It is hard to assess whether partial addition of a real mechanism (substitution in only a limited number of multifunctional situations) in a rather unrealistic manner (by assuming 100% substitution) will lead overall to more valid LCA results. As the outcome of the exercise is still virtually always a multifunctional system, some form of allocation procedure is still required. What might be said is that application of substitution to all the main processes might have the advantage of avoiding the arbitrariness of applying substitution in some cases and allocation in others.

Several LCA theorists, have developed proposals for handling market relations in LCA inventory modeling more satisfactorily. In particular Ekvall (1999) has developed methods for handling elasticities in LCA and Weidema (2001) has developed a substitution method which solves the problem of regress, where each 'substitution system' is itself multifunctional, requiring substitution, etc. A recent development that could no longer be included in this Guide but that deserves further assessment is the value-corrected substitution method, as described in Werner & Richter (2000) and Werner (2000). It expands the applicability of the substitution method to situations of recycling where the secondary material has a lower value than the primary. Then substitution is assumed not to be to the full amount but only a fraction, given by the ratio between secondary and primary price.

Ekvall, rather than unrealistically assuming full substitution (as in the avoided burden method) or zero substitution, advocates using realistic default values for elasticities of supply and demand. From a practical angle, Ekvall’s proposal brings with it the problem of massive system expansion, as ever more processes partially adjust to the changes in demand resulting from some choice introduced in the product system(s) under study. Weidema (2001), for his part, advocates adhering to ‘extreme’ elasticities which, while fairly unrealistic, can at least be handled systematically. His method assumes that there is one output from every multifunctional process, a change in demand for which will lead to full adjustment of the production volume of that process. In other words, supply of that product by the process is assumed to be fully elastic. Demand for all the co-products is then taken to be fully inelastic, so that depressed demand causes no decline in production but is offset by reduced production elsewhere in some other process delivering the same product. However, any increase in demand for the elastic product will lead to increased production of all the co-products. Combination with similar assumptions on the demand side leads to a number of extremes. If demand is fully inelastic, as is assumed for some near-waste products, extra supply will not reach the market and will become waste. If demand for the co-product is fully elastic - the other option - the extra co-product will fully replace production in some other process. On this rather unrealistic basis, Weidema defines a substitution procedure that can resolve the problem of endless regress, at least in some cases, by limiting the regress to two processes which symmetrically substitute for each other in ever smaller amounts. We hence refer to his variant of the substitution method as the ‘symmetrical substitution method’. It is not entirely clear how universally applicable this method is, nor if all the problems of multifunctionality can thus be resolved (for waste processing, for example). For a fuller description, we refer the reader to the refinery example elaborated by Weidema in Part 2b.

Assuming broad applicability for the moment, however, how are these two methods to be evaluated? The key problem with the symmetrical substitution method is that the techniques employed are of the market modeling type, while the specified market responses are not in line with available empirical

knowledge. However, it does solve the multifunctionality problem, at least to some extent. We therefore tend to regard Weidema's method not as part of inventory modeling, but as a type of allocation, i.e. as a procedure for resolving the partitioning problem defined by prior modeling. If a relatively simple and generally applicable allocation method is available, as we believe to be the case with economic allocation, there are no good reasons to opt for the complex and rather unrealistic method of Weidema. Ekvall attempts to be more realistic in the elasticities involved in substitution. For lack of a complete data set, however, here too simplifying assumptions must be introduced. If indeed applied broadly, an ever increasing number of processes would become part of the system analysed, still requiring a separate allocation step. Further research on how to operationally introduce a market modeling step such as that advocated by Ekvall (1999) is in itself interesting, but not for solving the multifunctionality problem. To gain an indication of the possible effects of substitution the symmetrical substitution method of Weidema may be used as a form of sensitivity analysis.

The other approach to incorporating market relations in LCA starts from general economics. As the marketing programmes of most firms show, there is a wealth of empirical economic knowledge available on market relations, backed up by increasingly sophisticated models. The general structure of these models is wholly compatible with the interests of industrial ecology, moreover, as they specify the overall adjustment at the systems level as induced by a specific technology or volume change. When it comes to their potential application in LCA, however, market models have two serious drawbacks. In the first place they require use of applied general equilibrium (AGE) models at a micro-level, while such models are still barely operational at an aggregated, meso-level. General application of market models at the level of technological detail required in LCA is consequently not yet feasible.

There is a second, equally fundamental problem. Increased demand for a given product (and process) never results in a full adjustment of supply. Because of the price rise induced by extra demand, demand for other goods and services will be depressed and their functions delivered to a lesser extent. This means that any switch to an alternative product will induce a virtually endless series of small changes in all other functions. Because of the price rise induced by extra demand, demand for other processes will be reduced, as will be the production volume of these other processes. The induced shift in volumes and prices is virtually endless. In each round an ever greater circle of processes will be influenced, involving ever more product systems, albeit affected to a diminishing extent. Although perhaps more realistic, this kind of analysis is not feasible as a method for comparing alternative product systems for equivalent functions, as in LCA. So why not skip LCA, then, and simply switch to this more realistic kind of market analysis, thereby abandoning this central restriction of LCA: the functional unit as the lynchpin of inventory calculations? There is a simple reason. Contemporary market analysis does not go into the kind of detail required for tying economic activities to environmental effects, for the complex task of accurately modeling all the various kinds of processes is beyond current data gathering and modeling capacity. The systems involved in market modeling generally span the globe, as does LCA, but now at a very high level of aggregation only.
The option of combining economic and environmental analysis in a non-LCA framework does, in principle, hold a certain appeal, though. If the key problem of evaluating combined changes in production and environmental effects could be solved, it might become a viable alternative to LCA. Bouman *et al*. (2000) have developed a stylised market model for use in LCA-type decision situations. With its rigorous simplifications, it is still a long way from being a realistic market model suitable for LCA-type questions. Experimental application of this market model and a typical LCA model to the same simplified case showed that the two approaches yielded very different conclusions. If market models, stylised and simplified as in LCA, could be applied more easily, the combined use of the two model types, each with their strengths and weaknesses, might be a better option than shifting from LCA to the market type of modeling.

As yet, market mechanisms have been incorporated only incidentally in inventory modeling, in the special case of substitution. If the assumption of fixed prices in the previous paragraph is removed considerable realism can again be added, as market processes are all around us. Including technical relations as well as process operators' aims in inventory models would provide an immediate specification of production functions. Again, for LCA purposes it is long-term technical relations, including long-term investments, that are relevant, and so it is long-term production functions that are relevant. Production functions specify the supply functions of the products involved. The combination of final demand and production functions specifies markets as combining supply and demand. Market

models are widely used, but not in LCA. The models used by economists are rarely technology-specific. Supply functions are based on technologies and on past investment decisions. If concrete knowledge is available, it often is confidential. Even without technological specifications, though, models are of only limited value as the complexity of market relations spirals with the number of processes involved. As a consequence, current operational models generally focus on a very restricted part of the system relevant in LCA; they are partial equilibrium models. Although economists have also developed more comprehensive Applied General Equilibrium (AGE) models, these are so aggregated that they have no part to play in the technology-specific decisions examined in LCA.

In what way, then, might incorporation of market mechanisms in inventory models contribute to LCA-type of analysis in the not too distant future? To reduce the attendant complexities, application might be restricted to just the main processes. When comparing high-speed trains and aircraft as two alternative modes of transport on a 500-km route, for example, market analysis could be used to determine the long-term supply and demand elasticities of each. Additional investments in railways would lead to a reduction in air ticket prices adequate to maintain a desired utilisation of adjusted air transport capacity. An expansion of rail transport capacity, by whatever mechanism, will not therefore lead to an equal reduction in air transport volume. Roughly speaking, the reduction in the latter would be about half the increase in actually used rail capacity, a midpoint between full substitution and no substitution at all. An increase in air transport capacity will have a similar but quantitatively different knock-on effect on rail transport volume. Car and bus transport volumes would also be affected, of course. In this way LCA could provide more realistic information for decision support on traffic modalities. At the same time, though, the analysis would also become far more complex. Assuming non-market inventory modeling to refer to single-function systems only (train or plane), the market-based analysis would show that besides a shift towards the extra train kilometres resulting from the investment decision in rail, air traffic would not be reduced by the same amount but substantially less. Overall transport would increase. After such an analysis it hardly seems sensible to allocate this extra air travel 'away' from the train system. This would effectively remove the extra information on the market mechanism.

A very different option for incorporating market dynamics is partial economic modeling, to set parameters for the system analysed, as Kandelaars (1999) has done (cf. 1.2.3.2). She gives the example of a policy-induced market shift from zinc gutters to PVC gutters, in which a new equilibrium is attained in the housing stock once all the old zinc gutters have been replaced. This kind of dynamic substitution can be included in the system model, with integration over time leading to the “average” inventory system. This is a deviation from standard LCA, as the functional unit is being supplied by a dynamic rather than steady-state system.

Beyond the realm of LCA, particularly for the purposes of energy analysis, larger models have been developed that not only incorporate technical (i.e. process) relations and the aims of process operators but also include dynamic path analysis, complex market mechanisms and/or macro-economic dynamics. One such model is the MARKAL model developed by ECN, which has been adapted for broad-brushstroke environmental analysis (Gielen *et al*., 1998; Seebregts *et al*. 1999). In principle, this model is also suitable for the decision-support domain of LCA. Its broad coverage of mechanisms constitutes its evident strength, but also its weakness: its complexity is such that practitioners must dispose over specialised modeling knowledge, for otherwise the model soon becomes an impenetrable black box. For larger-scale decisions, on future energy supply systems for example, it might nonetheless be a good option to include MARKAL-type models in the toolbox, not to replace but to augment LCA (or vice versa). These models provide no specific solution to the allocation problem, however. In highly aggregated versions in which each sector is assumed to produce just one product, the allocation problem does not arise. As such models are not particularly technology-specific either, they provide little scope for supporting the technology choices for which LCA is designed.

Finally, for the purpose of environmental decision support Cost-Benefit Analysis (CBA) can be applied, especially for investment decisions. Some CBAs may be formulated in terms of a functional unit, as when considering different options for expanding electricity production in CBA. It is usual but by no means necessary to weigh environmental effects in CBA by quantifying some measure relating to consumer preference. If this is not done, LCA impact assessment can be made into a separate chapter of CBA, leaving the economic part of it very similar to the inventory phase of LCA. It is now usual in CBA to specify the environmental aspects of the In this environmental sense CBA is not a systems analysis.

In principle CBA could be transformed into a systems analysis for environmental aspects as well if the upstream and downstream processes were specified at the detailed technological level then required. Market models are now used in CBA for such processes, but at an aggregate level only. Transforming CBA into an LCA-like systems analysis would involve the same problems as introducing market relations into LCA. The problems would be further compounded by another characteristic of CBA: its time-specific nature, at least for the main activities in the life cycle of the investment involved. Such a time-specific analysis cannot be incorporated in steady-state LCA as developed in this Guide. However, it is more compatible with the economic analysis for business decisions on product systems.

What can we conclude on incorporating market relations in LCA and how can this help solve allocation problems? Starting with the first question, the symmetrical substitution method of Weidema, still difficult to understand, is so unrealistic that it cannot be seen as a serious contribution to modeling, although technically it may solve the multifunctionality problem in a number of cases. The approach of Ekvall, while possibly more realistic, draws an ever larger number of processes into the system, as substitution is usually partial substitution. The use of market modeling to simulate shifts between product systems, as proposed by Kandelaars, may yield additional insight but does not touch on the multifunctionality problem itself. Larger models such as the MATTER/MARKAL models may have a role to play, but their complexity would seem to limit their general applicability to major studies on complex issues. Further work may allow market relations to be incorporated in LCA in some fashion, albeit at the cost of aggravating multifunctionality problems, which must then still be resolved in a separate allocation step.

Can market models help resolve or avoid the multifunctionality problem, then? Including substitution processes in inventory modeling is sometimes regarded as a solution, as a means of avoiding the multifunctionality problem altogether. This is indeed the case when the additional function(s) delivered by a combined production process can be substituted, fully, by (a) single cradle-to-gate system(s). Such cases are extremely rare, however. Growing application of the principles of industrial ecology will lead to ever greater process multifunctionality. In general, incorporating market mechanisms will cause a massive expansion of co-functions. In the example on transport, above, any capacity increase in one mode of transport to some degree influences all other modes. The system would therefore be multifunctional even if all the various functions involved were delivered entirely by monofunctional processes. As the vast majority of real-world processes are multifunctional, market modeling would increase multifunctionality to such an extent as to effectively preclude a solution through allocation. One solution would be to simply accept multifunctionality. As market prices are available in this economic context, the additional functions at the systems level could be expressed in monetary terms and aggregated into a single figure. This solution has the beauty of simplicity, in that it covers the sum total of additional functions in a comprehensive and comprehensible fashion. Its drawback, though, is that the resultant environmental profile is that of the combined system of target function and all the additional functions. To compare alternatives having different overall amounts of additional function in monetary terms, the share of the target function in this total must be established, and the only way to do so is to value the function in monetary terms, too. Its share in the total economic value of the system is then its share in the overall environmental burden, as specified. This, effectively, is economic allocation at the systems level. As argued presently, allocation at the process level is then to be preferred.

For practical and probably also theoretical reasons, then, full market modeling is not an option for the environmental decision support where LCA now is used for. Integrating real market modeling in LCA is even more difficult. Some additional market modeling, next to LCA, may be a best option, thus using a number of different models to indicate main mechanisms in the effects of choices.

Review conclusions

The first conclusion is that nobody advocates the formerly preferred options of allocation based on simple measures like mass or energy. The limited relevance of approaches grounded in ‘physical relationships’ is indeed explicitly stated in ISO 14041. At best, they can be used as a proxy for allocation based on economic value. There is also general agreement that allocation, if performed, should be done at the unit process rather than systems level. Many developments combine modeling adaptations and solutions to the multifunctionality problem. Keeping these steps apart would benefit modeling and would lead to clearer approaches to allocation.

As to improved specification of technical relations within processes, in the field of combined waste processing allocation procedures have been developed that distinguish between waste-specific emissions, reflecting physical causalities and hence belonging to the realm of modeling, and other process-related emissions, which cannot be related to specific inputs and which are allocated on a mass or energy basis, for example. The models employed partition some of the chemical elements in emissions to individual waste inflows according to the share of the latter in the total mass input of those elements. For the time being, and for these specific elements, this appears to be the best method available. However, it is at odds with the conventional thrust of causal modeling, which is based on marginal modeling, viz. varying the input and then seeing how the output varies. Also, this modeling approach may reflect short-term causal relations better than long-term. For the 'process-related emissions' (as distinguished by Eggels & van der Ven (1995); see above), which still need to be allocated to the different waste inflows, the solution chosen is more dubious and not in line with current notions of allocation. Inclusion of more specific causal mechanisms is to be regarded as a modeling improvement, preferably in tandem with inclusion of those long-term mechanisms relevant to LCA.

As to including the aims of process operators, this field is still largely open, although there is one exception. In combined (but not joint) processes, short-term optimisation modeling has been developed for systems operated by one owner. In this case it may indeed also help resolve the multifunctionality problem. However, short-term analysis is not generally that relevant to most LCA questions. Incorporating the aims of process operators and technical relations over the longer term in models remains an appealing goal that should be pursued further. Although proven for the short term, such optimisation methods require conceptual and empirical adjustment before they can be applied to the longer-term issues with which LCA is generally concerned. In most systems, furthermore, there will be a multitude of process operators. Straightforward optimisation is then not feasible. Optimisation in terms of minimising social costs seems more relevant to developing alternatives in LCA than as a modeling improvement or allocation procedure. The relationship between social cost assessment methods and LCA has been touched upon but not yet elaborated. Although optimisation models may someday be useful in applications in an LCA context, like environmentally specific market models they can by no means yet be defined as state of the art.

As to including market relations in LCA, attempts have not yet yielded operational methods. Starting from LCA, substitution, full or zero substitution is too far away from economic reality. It seems that introduction of such artificial and unrealistic modeling assumptions may indeed 'solve' some multifunctionality problems in a technical sense. As with other types of allocation, part of the multifunctional process then is subtracted from the full process. However, this should better not be seen as modeling, but as a specific means of allocation on the multifunctional model. As the additional insight gained is at the expense of unreal assumptions, it seems better not to use such methods in general practice but only additionally, in a sensitivity analysis, especially when the assumptions are not too far from reality.
If progress towards more realistic modeling in LCA is made, the problem of multifunctionality is not solved, not even partly, but increased. Thus, developments in the last decade have cleared the ground for improved modeling but not for a more elaborate solution to the multifunctionality problem. ISO options in terms of avoiding allocation and allocation based on physical causalities, when made operational, tend to become part of modeling, and do not then reduce multifunctionality. So the only remaining, more or less universally applicable option for allocating inputs and outputs among products (i.e. functions) is therefore to partition on the basis of their economic values, the main driving force behind economic processes. This option will be described in more detail below, and with operational detail in Part 2b of this Guide.

An alternative and more refined approach to allocation would be to use market prices that have been adjusted so as to incorporate negative environmental impacts and other social, or external costs. This option requires an adequate, i.e. social optimum regulatory regime, however, and is not elaborated here. Allocation would then not reflect the motives of process operators but objectives for society at large, including environmental objectives. Such a shift would mean a substantial deviation from the set-up of LCA, where environmental aspects are specified and evaluated independently from economic aspects.

PROSPECTS

As outlined, recent work has been concerned not so much with the specifics of allocation as with extending inventory modeling to overcome the traditional limitations of 'black-box' processes with fixed input-output coefficients, with subsequent implications for allocation constituting our prime interest here. In the coming years model refinement is likely to continue along the same three main lines of investigation, viz. inclusion of technical relations, aims of process operators, and market relations. Allocation based on economic value can be applied systematically and needs practical application to see how operational problems can be solved.

Further specification of technical relations is to be expected, one possible source of impetus being incorporation of some of the modeling tools used in engineering design. Most engineering models are concerned with short-term relations, however, based on short-term physical and other causalities. If such models could be extended to incorporate variations in installed capacities endogenously, they might be extremely useful in LCA. This kind of modeling tool could then effectively be used to examine alternative investment options. It would certainly reinvigorate the still rather open debate on which processes to include in the analysis, a choice essentially between marginal, non-constrained Including current state-of-the-art) processes. Knowledge demands would be enormous, however, and would oblige LCA practitioners to keep abreast of innovations in each and every process involved. In addition, investment functions are hardly a technical affair only, for investment decisions are based primarily on economic considerations. If such models are to be operationalised for use in LCA, therefore, they must be extended to include the aims of process operators. This also holds for combined waste treatment processes. Assuming (steady-state) functioning of such a process at some optimum level, long-term analysis of some additional amount of waste to be processed would indicate a need to install new capacity. Establishing the effects of processing the waste would then involve a comparison of the system flows with and without the functional unit. Economic considerations play a role in waste processing, too, albeit somewhat less well-defined. Here too, then, analysis shifts into the second group, with operators' aims being included in the inventory model. Overall, then, it is to be concluded that purely technical model specifications do not facilitate LCA modeling to any great degree. Even if such types of model became available for long-term analysis of waste management, they would still not resolve the allocation problem.

Incorporating the goals of process operators in models, as a first step towards market modeling, seems feasible in situations where one or just a few process operators dominate the system. Given some additional unit of function, the new optimum for the system could be specified, under the simplifying assumption of constant prices. Such models are quite common in operations research but are then concerned with short-term rather than long-term optimisation. Such optimisation models could to some extent resolve the multifunctionality problem, in cases of combined but not joint processes. Optimising from a collective point of view, as in minimising social cost, is not empirical modeling but may be useful in specifying attractive alternatives, which should be subsequently investigated by means of inventory modeling.

Incorporating the full complexity of market relations interlinking all the processes in the system is not operationally feasible at the level of technological detail required in LCA-type decision situations. If applied to the main processes only, it may become feasible. However, it doubtful whether allocation makes sense as a subsequent step, after modeling, as substantial information on market effects would then be lost. Without allocation we would leave the realm of functional unit-based LCA.

CONCLUSIONS

We now return to our original question: how to model the system and how to resolve the resultant multifunctionality problems?

In order to maintain a clear relation to ISO 14041 certain modeling aspects must be included, as can be seen above. The basic principles to be adopted in inventory modeling are summarised in Table 3.9.3. See further explanations in Sections 1.1, 2.3 and 3.1.

What starting points are there as requirements for allocation?

The first main principle, a basic requirement of the allocation procedure, is that the sum of the allocated values of all product flows equals the total value of the flows of the unallocated process: the '100% rule'.

Optimisation models do not generally satisfy this requirement. Allocation does not involve the principle of mass balance, which may hold for the unallocated, multifunctional process, but not for the monofunctional processes resulting from allocation. As most process descriptions do not include the consumption of oxygen or any other form of respiration, even unallocated processes do not in fact mass-balance. A second principle is that allocation should be at the level of multifunctional unit processes only. It is at that level that the clearest view is possible on technical relations, and on the values of the co-products.

A third, more methodological principle relates to reasonableness. In particular, results should be unaffected by the sequence of application of the allocation procedure (cf. Sen, 1969), i.e. which of the co-products is taken as the initial point of departure. Also, if processes are first allocated and then placed in the system scaled to the functional unit, the result should be the same as when they are first scaled, prior to the allocation step.

The fourth requirement is that the same principles should be applied to all the categories of co-production: combined and joint production, combined waste processing, re-use and recycling. The definition of the multifunction problem as modeled does not distinguish different types of multifunctionality, as the term 'product' covers both goods and services, allocation is necessary if more than one product is produced by a unit process. Waste having a negative value is not a product; nobody buys it. Processing waste is a service provided to the party supplying the waste, to be allocated to the products of the process producing them. That service relieves the producer of his waste; he pays for that service, being a product. Hence, the different situations for combined production, co-production, combined waste handling and recycling do not differ in their basic definition. In each case the same allocation principles should hence apply.

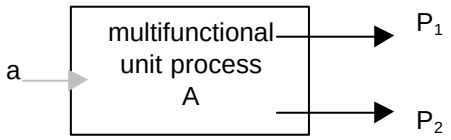
A fifth, more practical principle is that multifunctional processes associated with more than one of the product alternatives being compared should be allocated consistently. If different options for allocation are available, the same option should be applied to all the alternatives compared. Otherwise, any resultant differences between alternatives would be due both to real differences and to differences in the allocation methods applied. Of course in that situation a sensitivity analysis on the different options is due, to see how outcomes differ if one option for allocation is applied to all alternatives, or the other.

Table 3.9.3: Inventory modeling and allocation principles.

Resumé of modeling principles
– use fixed input-output coefficient process definitions wherever possible
– detail processes until a truly multifunctional core is reached (ISO 1a)
– include all processes implied in fulfilling the function
– define system boundaries where flows are being paid for by other product systems
– define system boundaries where environmental interventions enter or leave the system
– do not cut off flows that cannot be specified further but estimate them, e.g. using environmentally extended input-output analysis
– do not use market mechanisms in detailed LCA, but possibly in extended LCA
– make a clear distinction between system modeling and allocation
allocation principles
– '100% rule'
– allocate at the level of unit processes only
– independence of sequence of application
– consistent principles for all varieties of co-production
– consistent allocation procedure and results for multifunctional processes associated with different product alternatives

The guidelines for dealing with the multifunctionality problem have been elaborated primarily on the basis of economic allocation, in both an extended and a simplified version. These guidelines were co-developed with Lindeijer, and their background and more detailed formulations are elaborated in Appendix C. As the substitution method is regarded by some as a promising approach to the multifunctionality/allocation problem, we here present its best elaboration as an option for extended LCA. It does not constitute a recommended method, however, for the theoretical reasons discussed above and also because it is not entirely clear how operational choices for long-term change-oriented LCA are to be reasoned and made in the substitution approach. The method is illustrated with an example elaborated by Weidema for this Guide (see Part 2b, Section 3.9.3.2). Economic allocation is

defined as partitioning of all non-product inputs and outputs proportionally to the share of each product from the unit process in total proceeds, see Figure 3.9.1 below.



Allocation factor F is the share in total proceeds from sales of a product:
$F_1 = P_1 / P_1 + P_2$
a is the vector of all other inputs and outputs of the unit process

Figure 3.9.1: Economic allocation of a unit process as shares in its total proceeds

The steps in inventory modeling to avoid the multifunctionality problem, and the solutions to the multifunctionality problem through allocation are now surveyed. The framework for this survey is the two levels of LCA distinguished in this Guide, simplified and detailed analysis, with additional options for extensions. The latter are not in line with the starting points adopted here, however. The relationship of these steps to the framework of ISO 14041 is indicated, if they are included there. As ISO does not distinguish between simplified and a detailed LCA, some ISO elements are in either option, or ISO steps occur in different versions. Also, as ISO now does not fully distinguish between modeling as a possible cause of the multifunctionality problem and the available solutions to that problem, some steps are part of modeling in the Inventory analysis and some in the solution, through some sort of allocation. In due course such further differentiation might be incorporated in revisions of ISO 14040 and 14041.

The respective steps to be followed are summarised in general terms below, showing their equivalence, or otherwise, to the steps provided in ISO 14041 (
Table 3.9.4). The steps ‘economic allocation’, in detailed and simplified LCA, and ‘symmetrical substitution method’ (termed ‘avoiding allocation’ by Weidema, 2001), as an option for extension, are elaborated in greater detail in Volume 2b.

Table 3.9.4: Recommended procedure for handling multifunctionality in inventory modeling and allocation.

	detailed version	simplified version	options for extension
inventory modeling	**dm1**. Divide processes that are not really multifunctional into monofunctional unit processes (ISO step 1a) **dm2.** For waste processing: model long-term technical relations, assuming simple aims of process operators (ISO step 2, extended)	**sm1.** Use made-single databases (correspondence with ISO different per database and process) or use environmentally extended input-output data **sm2.** As a proxy, treat open-loop co-production as closed-loop co-production (after ISO), with quality adjustment **sm3.** Use available waste management models based on physical causality (ISO step 2, simplified)	**em1.** Apply market analysis to main co-products, as a realistic system expansion (after ISO step 1b, extended) **em2.** Set up a linear programming model of the main processes if operated by a single organisation (no ISO correspondence)
allocation	**da1.** Apply economic allocation based on market value or constructed market value (ISO step 3, interpreted economically)	**sa1.** Apply economic allocation where readily feasible, if not at unit process level then at full system level **sa2.** Use substitution with already made-single cradle-to-gate database data (ISO step 1b, simplified) **sa3.** Apply the most readily available physical product parameter related to value, e.g. mass, volume, energy content (not in line with ISO) **sa4.** For all remaining multifunctional processes: use some 'quick-and-dirty' measure: e.g. for recycling, '50% reduction of primary production' (not generally in line with ISO)	**ea1.** Apply the symmetrical substitution method (further interpretation and extension of ISO step 1b)

RESEARCH RECOMMENDATIONS

- Distinguishing more clearly between modeling and allocation opens new perspectives on improved inventory modeling.
- With this distinction clearer, the respective merits of LCA and other tools and models for empirical analysis of environmental impacts will be easier to clarify and hence improve.
- Modeling internal relations in processes from a long-term perspective is a challenge. This will require systems modeling to some degree and requires the explicit introduction of the aims of process operators.
- By incorporating certain economic mechanisms in the inventory model, particularly in cases involving extremely high or low elasticities, inventory modeling might be made more realistic and some of the principal defects of LCA redressed.
- Developing parallel models for decision support (e.g. LCA, market modeling, SFA, CBA), each indicating a key effect mechanism, is seen as a more fruitful route than incorporating ever more mechanisms in LCA.
- If such parallel modeling options become available, interpretation of results cannot be restricted to LCA but will involve the full set of available analytical tools.
- In this connection, research is specifically recommended on how LCA relates to Cost Benefit Analysis (CBA), which avoids allocation problems by accepting inter-system differences in function and expressing these in terms of monetary value. The relative merits of these two methods are as yet unclear.

- It is recommended, furthermore, to deepen understanding of the precise relationship between system boundaries and allocation, the definitions of which are closely related;
- Research is also recommended on how substitution can be more systematically integrated in inventory modeling than is currently feasible.

3.10 Calculation method

TOPIC

Collection of process data yields a database of processes. The act of quantitatively relating these processes to one another, scaled to the reference flow following from the functional unit, is referred to here as the calculation method. The calculation result is a set of linked and scaled processes, each with scaled environmental interventions, which are usually aggregated.

DEVELOPMENTS IN THE LAST DECADE

In ISO 14041 (1998E) several steps are distinguished under the heading of calculation procedures:

- validation of data;
- relating data to the unit process;
- relating data to functional unit and data aggregation;
- refining the system boundaries.

Heijungs *et al.* (1992)
In the 1992 guide scaling of process data, there termed "Creating the inventory table", consisted of two steps (pp. 37–40): "quantification of the environmental interventions" and "representation of the qualitative environmental interventions".
Two calculation procedures were discussed in the 1992 Backgrounds document: the sequential method and the matrix method. The first is still widely used. However, with this method there is no quick and accurate way of dealing with processes which are mutually related (recursion). The matrix method allows feedback to be dealt with immediately.

The remarks on data validation in ISO 14041 have already been cited in Section 3.5. Relating data to unit process has been discussed in this Guide under the heading of data collection (Section 3.6).

On relating data to the functional unit and data aggregation ISO 14041 clause 6.4.4 states::

Based on the flow chart and system boundaries, unit processes are interconnected to allow calculations on the complete system. This is accomplished by normalizing the flows of all unit processes in the system to the functional unit. The calculation should result in all system input and output data being referenced to the functional unit.
Care should be taken when aggregating the inputs and outputs in the product system. The level of aggregation should be sufficient to satisfy the goal of the study. Data categories should only be aggregated if they are related to equivalent substances and to similar environmental impacts. If more detailed aggregation rules are required, they should be justified in the goal-and-scope-definition phase of the study or should be left to a subsequent impact-assessment phase.

Source: ISO 14041, 1998E

ISO 14041 (1998E) clause 6.4.1 states (see textbox):

When determining the elementary flows associated with production of electricity, account shall be taken of the production mix and the efficiencies of combustion, conversion, transmission and distribution. The assumptions made shall be clearly stated and justified. Whenever possible, the actual production mix should be used in order to reflect the various types of fuel that are consumed.
Inputs and outputs related to a combustible material, e.g. oil, gas or coal, can be transformed into an energy input or output by multiplying it by the relevant heat of combustion. In this case it shall be reported if the higher heating value or the lower value is used. The same calculation procedure should be consistently applied throughout the study.
[...]. All calculation procedures shall be explicitly documented.

Source: ISO 14041, 1998E.

A distinction between aggregated and non-aggregated inventory results may be useful. In the "Contribution analysis" part of the Interpretation phase, one of the recommendation is to determine the contribution of each process to the total inventory results, for instance. This implies that options enabling such analysis must be provided in the calculation step.

ISO sets no restrictions on calculation methods. Currently available methods include matrix inversion (Heijungs *et al*.,1992; Möller, 1992; Heijungs & Frischknecht, 1998), (simultaneous) sequential calculation of the inputs and outputs of each unit process of the system, with or without a number of iterations, and linear programming. As most LCA studies to date have not specified the calculation methods employed, there may be even more methods in use.
The main difference between these different calculation methods concerns the handling of loops, e.g. in the (frequently occurring) case that coal is needed for electricity production and electricity is needed for the extraction of coal. Such loops can be handled by matrix inversion techniques, but cannot be dealt with appropriately by most sequential calculation methods. In the latter the simplification is made to cut out loops altogether, by setting certain input flows to zero.[1] Although this is a very common approach, it is generally left implicit.
LCA studies are generally performed using dedicated LCA software programs. Several programs are available; for a software review see e.g. Rice (1996), Rice *et al*. (1997), Menke *et al*. (1996) and Siegenthaler *et al*. (1997).
The ISO text on refining the system boundaries focuses on sensitivity analysis and may be regarded as an issue for Interpretation, here treated in Section 5.6.
The result of the calculation step is termed the inventory table, which comprises all the environmental interventions related to the reference flow (of reference process) specified in the goal and scope of the study (see Figure 2.4.1). Furthermore, all economic flows not followed to the system boundary should be reported directly below the inventory table.

PROSPECTS
As mentioned in Section 3.6, LCA software should preferably allow scaling of data, including a time dimension. On this topic no further specific developments are foreseen.

CONCLUSIONS
We conclude that the best available practice with respect to calculation methods is matrix inversion, although this is not included in most software. If a different calculation method is employed, its precise nature should be specified as well as the differences from matrix inversion.

RESEARCH RECOMMENDATIONS
– LCA software should preferably allow scaling of data, including a time dimension.

[1] Sequential calculation techniques can, in principle, handle loops by iterating a given number of times, but most LCA software programs do not include this type of sequential calculation method.

4. Impact assessment

4.1 General introduction

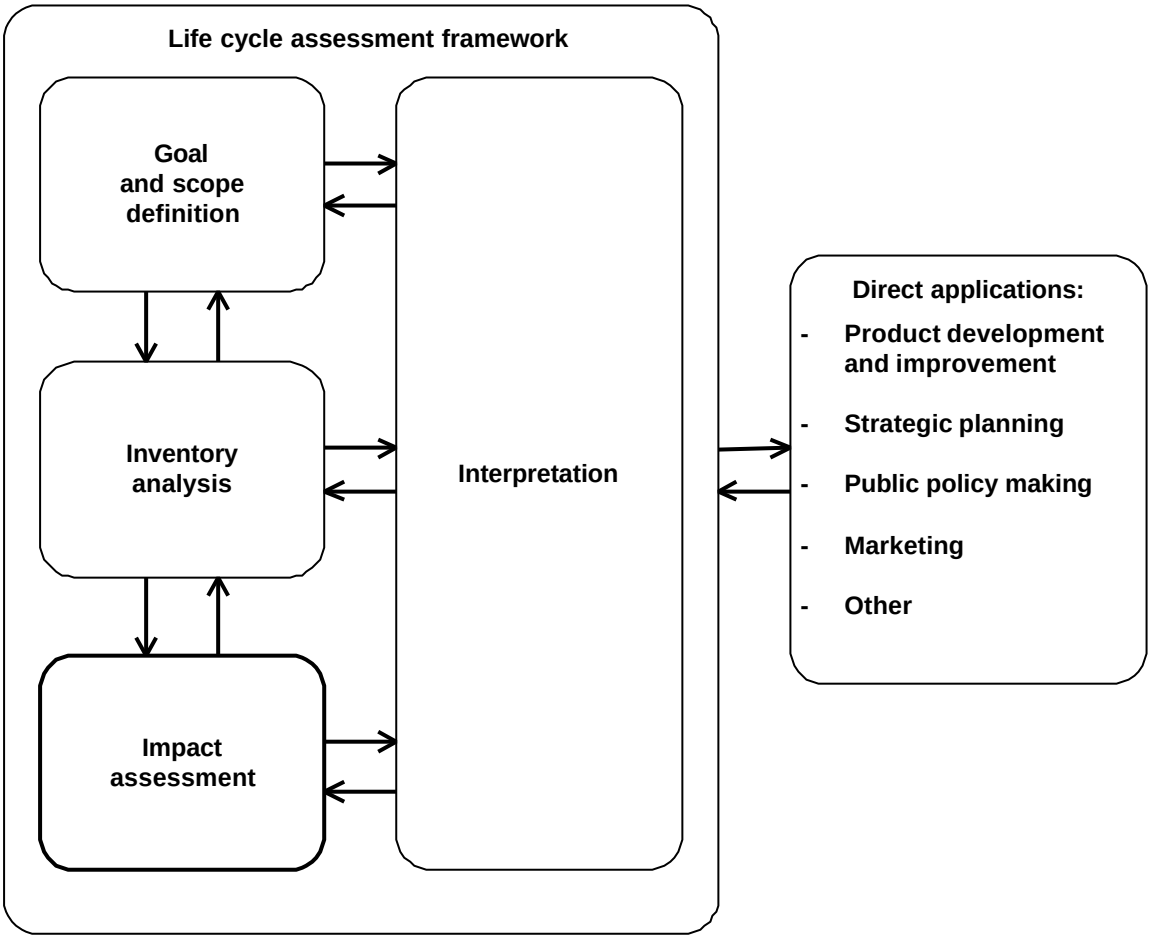


Figure 4.1.1**:** The Impact assessment phase as part of the general methodological framework of LCA (source: ISO 14040, 1997E).

According to ISO 14040 (1997E) Life Cycle Impact Assessment (LCIA), the third phase of life cycle assessment, "is aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts of a product system". In a similar vein, the first SETAC-Europe Working Group on Impact assessment (WIA–1) defines LCIA as "a quantitative and/or qualitative process to identify, characterise and assess the potential impacts of the environmental interventions identified in the Inventory analysis" (Udo de Haes (ed.), 1996). To this end the individual data of the inventory table, or LCI results, are translated into contributions to selected impact categories, such as 'depletion of abiotic resources', 'climate change' or 'acidification'. These contributions are calculated using characterisation models, in which relevant environmental processes are modeled to a so-called category endpoint. To aggregate the results for each category, these are expressed in terms of a common parameter called a category indicator: 'infrared radiative forcing' for climate change, for example.
ISO 14042 (2000E) puts it as follows: "The LCIA phase models selected environmental issues, called impact categories, and uses category indicators to condense and explain the LCI results. Category indicators are intended to reflect the aggregate emissions or resource use for each impact category. These category indicators represent the 'potential environmental impacts' discussed in ISO 14040. In addition, LCIA prepares for the life cycle Interpretation phase". (See Figure 4.1.1). ISO goes on to note

that LCIA can assist in various decision situations, but that "parties should recognise that a complete product system assessment is difficult and may require the use of several different environmental assessment techniques". The latter means that it may sometimes be useful to apply other tools in addition to LC(I)A (see also Sections 2.4 and 2.5 of Part 1 of this Guide and Appendix B of this Part).

As elaborated in this Guide, Impact assessment comprises eight steps (see Section 1.4):
- Procedures (no special section in this Part; see Chapter 1);
- Selection of impact categories (Section 4.2, p. 141);
- Selection of characterisation methods: category indicators, characterisation models (Section 4.3, p. 150);
- Classification (Section 4.4, p. 241);
- Characterisation (Section 4.5, p. 243);
- Normalisation (Section 4.6, p. 244);
- Grouping (Section 4.7, p. 247);
- Weighting (Section 4.8, p. 249).

In elaborating these Impact assessment steps a further point of departure was ISO 14042 (2000E), with respect to the methodological framework and the issues for Impact assessment proposed there. Here the ISO proposals have been further operationalised taking into account the work of WIA–1 (Udo de Haes (ed.), 1996 and Udo de Haes *et al*., 1999) and relevant proposals by other authors. Deviations from ISO have been introduced only in cases where there are significant arguments for doing so.

In discussing the substance of the eight steps as well as the individual impact categories themselves, a fixed format has generally been employed below: 'Topic', 'Developments in the last decade', 'Prospects', 'Conclusions' and 'Research recommendations'. Where this format was less appropriate, as with the rubrics 'Interventions ' and 'Economic flows not followed to system boundary', it has not been strictly adhered to. As explained in Section 1.5 the step dealing with procedures is not discussed separately in this chapter, but in an integrated manner for all the various phases in Section 1.3.

We first provide a short resumé of the international organisations involved in work on Impact assessment, the main ISO requirements and the relation between ISO and SETAC, as a springboard for discussing the substance of the individual steps of LCIA.

4.1.1 International organisations involved

Since 1992, two organisation in particular have been involved in work on LCIA:
- ISO;
- SETAC, including specifically the SETAC-Europe Working Group on Impact assessment (Udo de Haes (ed.), 1996) and the SETAC-US Work Group on Impact assessment (Barnthouse *et al*., 1997).

In 1993 the International Organization for Standardization (ISO) established a technical committee (TC 207) concerned with standardisation of a number of environmental management tools, including a subcommittee on LCA (SC5). One of the Working Groups formed under SC5 has dealt with LCIA (ISO/TC 207/SC 5 /WG 4). The main tasks were: to define concepts, to define a technical framework for LCIA and to specify general methodological requirements and procedural requirements. These include requirements for 'comparative assertions disclosed to the public', which are much stricter than for other (either internal or non-comparative) applications. This work resulted in the International standard on LCIA (ISO 14042, 2000E).

Through its North American and European branches, the Society of Environmental Toxicology and Chemistry (SETAC) has played a leading role in bringing LCA practitioners, users and methodology developers together to collaborate on the continuous improvement and harmonisation of LCA methodology. SETAC focuses on the scientific development of LCA methodology. In 1993 a 'Code of Practice' was published (Consoli *et al*., 1993). From then on specific activities were taken up by a series of working groups, including the SETAC-Europe Working Group on Impact assessment (WIA) and the SETAC North America Work Group on Impact assessment. Initial reports have meanwhile been published independently (Udo de Haes (ed.), 1996; Barnthouse *et al*., 1997).

The European report 'Towards a methodology for Life Cycle Impact assessment' (Udo de Haes (ed.), 1996) provides an up-to-date, comprehensive review and critical analysis of existing thinking on LCIA. The principal methodological choices made in each phase of LCIA are analysed. The successor to this first WIA (WIA–2) has already started its work, again in Europe. The goal of WIA–2 is to establish a list of recommended impact categories, together with category indicators to be used in LCIA. This list should, as far as possible, be in line with ISO 14042, which, according to WIA–2, means it should have

maximum scientific and technical validity. Furthermore, it should be practicable in terms of number of categories, characterisation methods and inventory data requirements (Udo de Haes *et al*., 1999). The North American report is principally a critical review of the LCIA framework, scrutinising the accuracy of the various methods available and their applicability in various circumstances. Both the WIA and the SETAC North America Work Group have made significant inputs to the ISO process vis-à-vis development of an LCIA standard.

A major new task is the cooperation between SETAC and UNEP on establishing 'best available practice' in the field of life cycle Impact assessment, as a follow-up to the work of WIA–2. More specifically, such practice is to be established with respect to (a list of) impact categories, and category indicators and characterisation factors for each of these. It is anticipated that an appropriate form of cooperative organisation will be defined for this task in the year 2001 and possibly also established.

Besides these international developments and standardisation activities, there have been numerous proposals for defining impact categories, category indicators and models; these are discussed in Sections 4.2 and 4.3.

4.1.2 ISO 14042 requirements

ISO 14042 (2000E) describes procedures rather than specific methodologies or models for life cycle Impact assessment, implying that any methodology or model is acceptable as long as it satisfies the general ISO criteria. Figure 4.1.2.1 summarises the overall framework of LCIA, showing the relationship between life cycle inventory results, impact categories, category indicators and category endpoint(s), and illustrating these concepts with reference to the impact category 'Acidification'.

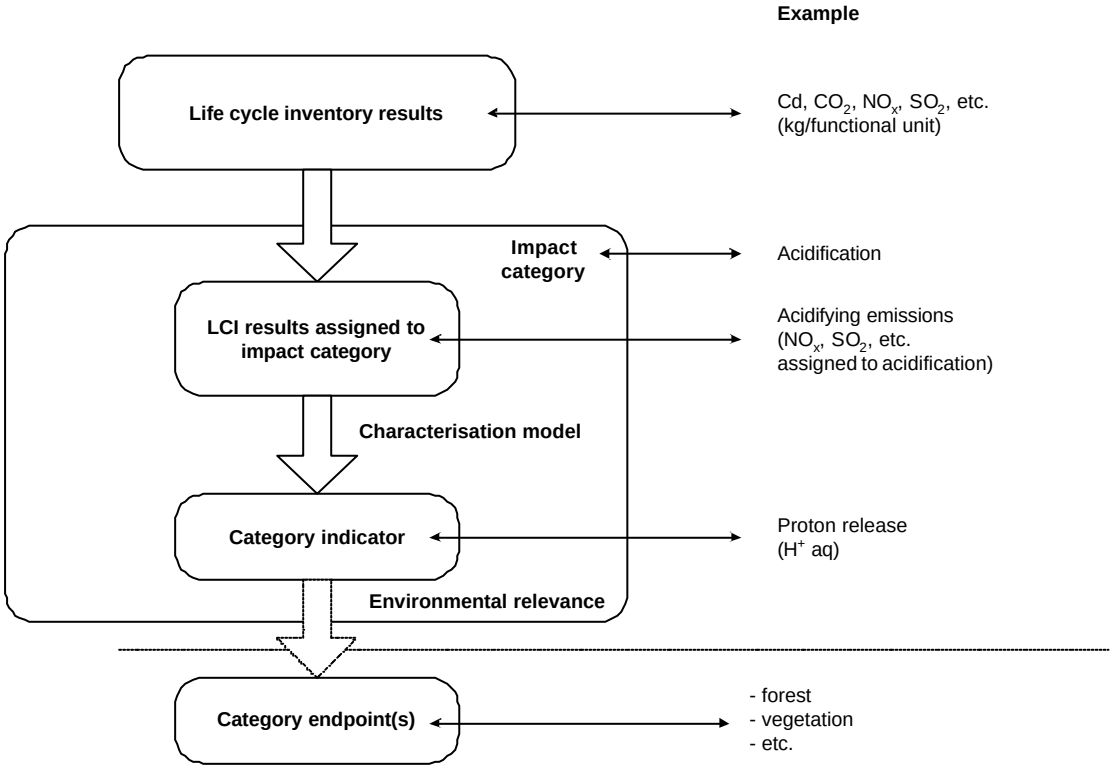


Figure 4.1.2.1: The conceptual framework for defining category indicators (slightly adapted from: ISO 14042, 2000E).

ISO 14042 also defines the term 'environmental mechanism': "a system of physical, chemical and biological processes for a given impact category, linking the LCI results to category indicators and to category endpoints". In this Guide we have opted to refine this definition, to distinguish more categorically between the terms of the 'real world' and those of the characterisation models used to

simulate certain partially underststood environmental processes.[1] Thus, the term 'environmental process' has been adopted here for the chain of physical, chemical and biological events in the natural environment that link a particular environmental intervention to a particular impact; typical examples include pollutant accumulation or leaching. For a given impact category, these environmental processes then together form the 'environmental mechanism', which is modeled to a greater or lesser extent by the characterisation model, up to one or more category endpoints.

According to ISO 14042, category indicators may be chosen anywhere along the environmental mechanism between intervention and endpoint. Operationalisation of each of the chosen impact categories then comprises the following elements:

- identification of one or more category endpoints;
- definition of a category indicator for each of these endpoints;
- identification of the LCI results to be assigned to each category indicator, taking into account the selected category endpoint(s); and
- identification of the characterisation model and characterisation factors to be used.

ISO 14042 states that this procedure "facilitates the collection, assignment, and modeling of appropriate LCI results" and "helps to highlight the scientific and technical validity, assumptions, value-choices and degree of accuracy in the model".

[1] In preparing this Guide the ISO terminology proved to be rather inconsistent on a variety of points and for this reason we have here refined the definitions of certain ISO terms. In the Glossary these are specifically indicated.

With respect to the selection of impact categories, category indicators and models, ISO sets the following requirements (see textbox):

a) the selection of impact categories, indicators and models shall be consistent with the goal and scope of the LCA study
b) the sources for impact categories, indicators and models shall be referenced;
c) the selection of impact categories, indicators and models shall be justified;
d) accurate and descriptive names shall be provided for the impact categories and category indicators;
e) the selection of impact categories shall reflect a comprehensive set of environmental issues related to the product system being studied taking the goal and scope into consideration;
f) the environmental mechanism and model which relate the LCI results and indicator as a basis for characterisation factors shall be described;
g) the appropriateness of the use of the characterisation model for deriving the category indicator in context of the goal and scope of the study shall be described.

In addition a number of recommendations is given for the selection of impact categories, indicators and models:

a) the impact categories, indicators and models should be internationally accepted i.e. based on an international agreement or approved by an international body;
b) the impact categories should represent the aggregated emissions or resource use of the product system on the category endpoint(s) through the indicators;
c) value-choices and assumptions made during the selection of impact categories, category indicators and characterisation models should be minimised;
d) the impact categories, category indicators and characterisation models should avoid double counting unless required by the Goal and scope definition, for example when the study includes both human health and carcinogenicity;
e) the characterisation model for each category indicator should be scientifically and technically valid, and based upon a distinct identifiable environmental mechanism and/or reproducible empirical observation;
f) the impact categories and indicators should be environmentally relevant;
g) it should be identified to what extent the characterisation model and the characterisation factors are scientifically and technically valid.

Depending on the environmental mechanism and the goal and scope, spatial and temporal differentiation of the characterisation model relating the LCI results to the indicator should be considered. The fate and transport of the substances should be part of the model.
LCI results other than mass and energy flow data included in an LCA study, e.g. land use, shall be identified and their relationship to corresponding indicators shall be determined.

The environmental relevance of the category indicator or characterisation model should be clearly stated in terms of the following criteria:

a) the ability of the category indicator to reflect the consequences of the LCI results in the category endpoint(s) at least qualitatively;
b) the addition of environmental data or information to the characterisation model with respect to the category endpoint (S), including
    - the condition of the category endpoint(s),
    - the relative magnitude of the assessed change in the category endpoints,
    - the spatial aspects, such as the area and scale,
    - the temporal aspects, such as duration, residence time, persistence, timing, etc.,
    - the reversibility of the environmental mechanism, and
    - the uncertainty of the linkages between the characterisation model and the changes in the category endpoints.

Source: ISO 14042 (2000E).

4.1.3 Relation between ISO and SETAC Working Group on LCIA

Since ISO 14042 describes no specific methodologies or models for use in Impact assessment, there is a need to specify LCIA methodologies and models that satisfy the broader ISO requirements. This is where the work of the SETAC-Europe Working Group on Impact assessment comes in, for WIA–2 is seeking to draw up an authorised, recommended list of impact categories complete with category indicators and characterisation factors. Proceeding from the ISO requirements, WIA–2 aims to establish a best available practical method for each impact category distinguished. At the time of writing, the work had yielded a list of recommendations for individual impact categories as well as an overall framework for these categories (Udo de Haes *et al*., 1999).

As stated in Chapter 1, the aim of this new Guide is to operationalise the ISO standards, and in particular ISO 14042, by updating and expanding the Guide of Heijungs *et al*. (1992) to incorporate all relevant developments since publication of the latter, taking the ISO standards as the basic point of departure and with particular reference to ongoing work within the SETAC community. For the LCIA phase, this means that the work of WIA–2 (Udo de Haes *et al*., 1999) has been taken as a starting point. WIA–2 was established in the knowledge that its mission would be taken over at the global level by the scheduled cooperation between SETAC and UNEP. The WIA–2 work on best available practice has not yet been completed and this Guide can therefore do no more than make recommendations on best available practice as understood at the present time. Thus, the present Guide goes beyond the work of WIA–2, taking into account as far as possible all relevant developments in LCIA during the last decade.

4.2 Selection of impact categories

TOPIC

In the Impact assessment phase the results of the Inventory analysis are translated into contributions to relevant impact categories, such as depletion of abiotic resources, climate change, acidification, etc. To this end, relevant impact categories must be identified. The text of ISO 14042 might be interpreted as indicating that these impact categories are to be defined anew for each study. To facilitate the work of practitioners a default list of impact categories has here been elaborated, thereby distinguishing between 'baseline' impact categories, 'study-specific' impact categories and 'other' impact categories. In this step of the LCA, then, practitioners are still obliged to select those categories relevant to the goal of his or her particular study, supported by the preliminary selection made in this Guide.

DEVELOPMENTS IN THE LAST DECADE

ISO 14042 does not provide a default list of impact categories for inclusion in LCIA.

The starting points of ISO and WIA–2 regarding the selection and definition of impact categories are summarised in Table 4.2.1. For the underlying argumentation the reader is referred to ISO 14042 and Udo de Haes *et al*. (1999), respectively.[1]

[1] Furthermore, ISO sets several requirements regarding the description and documentation of categories.

Table 4.2.1: Starting points for definition and selection of impact categories in general and for a specific LCA study.

General starting point for the framework of impact categories and category indicators: a framework shall be developed which is open to further scientific progress and further detailing of information (WIA–2)
General starting points for the total set of impact categories: 1. the categories shall together permit an all-encompassing assessment of relevant impacts, as currently understood (completeness) (ISO/WIA–2) 2. the categories should have minimum overlap and avoid double counting unless so required by the goal and scope (ISO/WIA–2) 3. the categories should be internationally accepted, i.e. based on an international agreement or approved by a competent international body (ISO) 4. the total number of impact categories should not be too high (WIA–2)
Starting points for the selection of categories in a specific LCA study: 1. the selected impact categories shall be consistent with the goal and scope of the LCA study (ISO) 2. the selected impact categories shall form a comprehensive set of environmental issues related to the goal and scope of the LCA study (ISO)

Choice of overall impact assesment method

Within the ISO framework the environmental profile resulting from the characterisation step is an important LCA result in its own right, with the grouping and weighting steps (which are more value-based) constituting distinct, optional elements. Some existing methods deviate in this respect, however, weighting interventions directly. One example is the Ecopoints method, in which emissions and extractions are weighted using a distance-to-target method, i.e. based on policy targets (Ahbe *et al*., 1990). Consequently, this method does not include a separate characterisation step.

A second major defining aspect of Impact assessment methods is the point in the environmental mechanism at which the category indicators are defined. They may be defined close to the intervention (the midpoint, or problem-oriented approach, e.g. Heijungs *et al*., 1992; Udo de Haes (ed.), 1996; Haas, 1997; Wenzel *et al*., 1997; Beetstra, 1998; Udo de Haes *et al*., 1999). Alternatively, they may be defined at the level of category endpoints (the endpoint, or damage approach, e.g. EPS: Steen & Ryding, 1992; Steen, 1996; ExternE: EC, 1995a; Eco-indicator 99: Goedkoop & Spriensma, 1999). A cluster of category endpoints of recognisable value to society is referred to as an “area of protection”. Here, we distinguish four: human health, natural resources, the natural environment and the man-made environment.

Despite its name, the midpoint approach still allows definition of category indicators anywhere along the environmental mechanism in question (cf. Udo de Haes *et al*., 1999), including the endpoint level. This permits use of the best indicator available for each impact category, regardless of where it is located in the environmental mechanism. If indicators are chosen at midpoint level their relationship with the category endpoints should be clearly defined. This will generally be in qualitative terms.

In ISO 14042 (2000E) the degree of linkage between the chosen category indicator and the category endpoint is referred to as the “environmental relevance” of the indicator. The environmental relevance is to be clearly stated in terms of the following criteria:

1. the ability of the category indicator to reflect the consequences of the LCI results on the category endpoint(s), at least qualitatively;
2. the addition of environmental data or information to the characterisation model with respect to the category endpoint(s), including
   - the condition of the category endpoint(s),
   - the relative magnitude of the assessed change in the category endpoint(s),
   - the spatial aspects, such as area and scale,
   - the temporal aspects, such as duration, residence time, persistence, timing, etc.,
   - the reversibility of the environmental mechanism, and
   - the uncertainty of the linkages between the characterisation model and the changes in the category endpoints (ISO 14042, 2000E).

In the second case category indicators are defined at endpoint level. The advantage of this approach is that the environmental relevance of the category indicators is high: it is this level which ultimately matters to society and which enables a direct link to be made with weighting methods. However, these kinds of endpoint indicators are still under development and the associated models do not yet include all the relevant effects of common interventions (see textbox below).

Since 1992 considerable efforts have been devoted to developing models and category indicators based on the endpoint approach (e.g. EC, 1995a; Goedkoop & Spriensma, 1999; Müller-Wenk, 1997; Hofstetter, 1998). The most comprehensive and recent work in this area is the Eco-indicator 99 by Goedkoop & Spriensma (1999). This approach is reviewed in the textbox and compared with the midpoint, i.e. problem-oriented approach.

In the Netherlands two Impact assessment methods have been developed in the last decade, both grounded in the 'environmental themes' formulated by the Dutch Government in 1989 (VROM, 1989; RIVM, 1991). Both have the same basic structure, with the indicator results obtained by multiplying the inventory results by the appropriate characterisation factor together forming the so-called environmental profile, which is then normalised (see section 4.6), before serving as input for a possible weighting step. Where the two methods vary is with respect to the characterisation models and characterisation factors developed and proposed for the individual themes.

In terms of their operationalisation there are also several clear differences between the methods. The first method, often referred to as the problem-oriented approach and first presented by Heijungs *et al.* (1992), operationalised models and characterisation factors for a number of impact categories, but did not operationalise the weighting step.

The second Dutch method is the Eco-indicator approach, developed primarily for the purposes of 'eco-design'. Designers were deemed unable to work with 10–20 indicator results, and the Eco-indicator therefore employs only 1 to 3 weighted indices. Thus, there is greater emphasis on weighting than in the approach of Heijungs *et al* . In the first version of the Eco-indicator (Eco-indicator 95; Goedkoop, 1995) weighting was based partly on a damage approach, partly on a distance-to-target approach (i.e. based on predefined damage targets). Most of the impact categories identified were adopted from Heijungs *et al*., although the two toxicity themes were defined rather more narrowly.

Originally conceived as an experiment, the Eco-indicator method has since been improved. In the latest version (Eco-indicator 99; Goedkoop & Spriensma, 1999) a completely different approach to Impact assessment has been adopted in which a limited number of damage categories are weighted (by a panel, for example). Three types of damage are distinguished, for which weighting is taken to be more readily feasible:

- damage to resources;
- damage to ecosystem quality;
- damage to human health.

As in the problem-oriented approach, the natural sciences are used to calculate the relation between the impacts of a (product) system's life cycle and the resultant damages. The Eco-indicator methodology thus consists of two parts:

- scientific calculation of the three forms of damage due to the life cycle of the product under study;
- a valuation procedure to establish the significance of these damages.

The method has a modular structure (Figure 4.2.1; source: Goedkoop, 1997) in which the building blocks of the natural science component can be modified or replaced to reflect different value systems (viz. Egalitarian, Individualist, Hierarchist). The authors recommend using the Hierarchist version of the model as the default method, with the other two being run as a form of sensitivity analysis (Goedkoop & Spriensma, 1999).

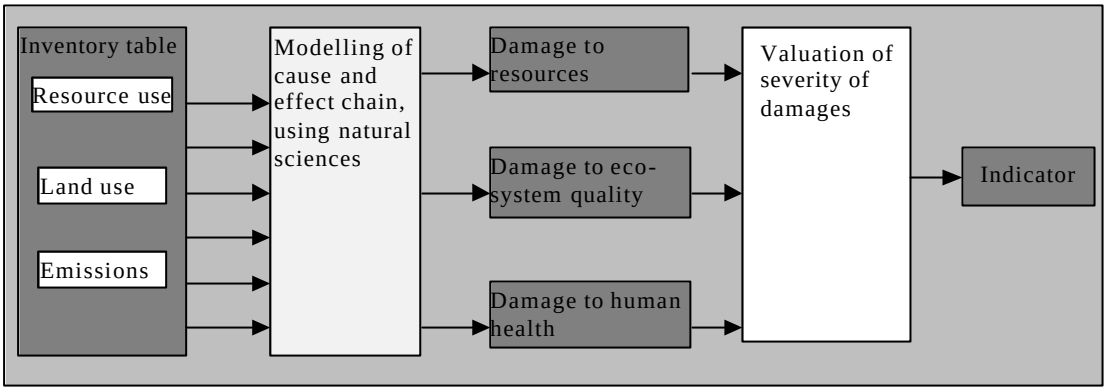


Figure 4.2.1: The modules of the Eco-indicator 99 method (source: Goedkoop, 1997).

In the Eco-indicator 99 approach, 'damage to health' is operationalised using the notion of DALYs: Disability-Adjusted Life Years. This indicator is said to measure "the total amount of ill health, due to disability and premature death, attributable to specific diseases and injuries. The DALY concept thus compares time lived with disability (YLD: Years Lived Disabled) ands time lost due to premature mortality (YLL: Years of Life Lost). Health is simply added across individuals. That is, two people each losing 10 years of disability-free life are treated as the same loss as one person losing 20 years" (Goedkoop & Spriensma, 1999). For the technical details of the DALY concept, the reader is referred to (Box 4.1) of this publication.
Both the problem-oriented approach and the Eco-indicator approach conform to the ISO 14042 (2000E) framework, as reflected in ISO/TS 14047 (*in prep.*), since both clearly distinguish the characterisation and weighting steps.

Although the Eco-indicator 99 approach is very promising and certainly appealing as an avenue for further research, the problem-oriented approach is currently considered the 'best available practice' for Impact assessment and has therefore been adopted in this Guide. The Eco-indicator method still has several serious shortcomings. It includes far fewer inventory items and provides only very limited coverage of human-toxic impacts (carcinogeneity only, thus ignoring a wide range of other health impacts). Some of the constituent models are outdated compared with those now used in the problem-oriented approach, while others involve major uncertainties. Thus, the terrestrial acidification and eutrophication models are based on the local, Dutch situation, while the problem-oriented approach now uses a European model (Huijbregts, 1999b); the data and assumptions of the toxicity model can be improved (*cf.* Huijbregts 1999a); and linkage of GWP and ODP and other universally accepted factors to damage parameters is still very incomplete and uncertain. Finally, aggregation of ecotoxicological impacts with eutrophying, acidifying and land use impacts is still very preliminary, and the ecosystem impacts of climate change, increased UV radiation and photochemical smog are not yet included.
The key feature of the problem-oriented approach is that the category indicators are defined at midpoints along the environmental mechanism, congruent with current environmental policy themes, and can therefore be modeled relatively accurately. The approach has the added advantage of permitting flexible choice of characterisation model and position of category indicator in the environmental mechanism, since for many impact categories more than one model is defensible and available. However, midpoints (wherever their precise position in the mechanism) are a difficult input for weighting and in the problem-oriented approach there is thus still no set of weighting factors covering all impact categories (see section 4.8).
The main advantage of the Eco-indicator 99 is that category indicators are defined at the endpoint level, giving them greater environmental relevance. As it is this level that ultimately matters to society, the object of the weighting procedure is more immediate. The major uncertainties associated with modeling from midpoints to endpoints constitute a serious drawback, however.
It would be very useful to collaborate and examine whether the advantages of the Eco-indicator 99 and the problem-oriented approaches can be combined into a still better and more comprehensive Impact assessment methodology.

Although the Eco-indicator 99 approach is very promising and is certainly appealing as an avenue for further research, the problem-oriented approach is currently deemed the 'best practice' for Impact assessment and has therefore been adopted in this Guide. As discussed in the text box, the former approach does not employ reliable endpoint indicators for all relevant impacts and has a number of other shortcomings, and cannot yet therefore be used to generate a comprehensive environmental profile. As things stand, therefore, we here recommend the problem-oriented approach, with impact categories defined at the midpoint level. This allows the best available indicator to be used for each impact category, regardless of where in the environmental mechanism between intervention and endpoint this category indicator is defined. In the future, when more complete endpoint indicators are available, an approach based on categories defined at endpoint level, such as the Eco-indicator 99, may well become the preferred approach (see also textbox).

In this Guide, the problem-oriented (midpoint) approach will now be elaborated further as a baseline. The Eco-indicator 99 approach can be used for the purpose of sensitivity analysis, since it is presently the most comprehensive method based entirely on endpoint indicators. The models used to calculate the indicator results for the various impact categories are more up to date and complete than comparable foreign approaches such as the ExternE and Environmental Priority Strategies, or EPS, methods (EC, 1995a; Steen, 1993 & 1996[1]). By using the Eco-indicator 99 approach as a sensitivity analysis, LCA practitioners can familiarise themselves with the kind of results yielded by this approach and compare them with the results obtained with the problem-oriented approach adopted in this Guide as a baseline.

Choice of impact categories

In elaborating the problem-oriented approach, a default list of impact categories first needs to be defined. Table 4.2.2 presents such a list, based mainly on the work of the WIA–2 Working Group on Impact assessment and earlier work (Udo de Haes (ed.), 1996; Udo de Haes *et al.*, 1999)[2], on more recent developments in the LCA field and, of course, on the basic starting points of Table 4.2.1. This default list acknowledges three groups of impact category:

Group A: 'Baseline impact categories' comprises those of the categories distinguished and discussed in Udo de Haes *et al.* (1999) for which a baseline characterisation method[3] is selected below, in Section 4.3. Group A impact categories are included in almost all LCA studies. Two revisions have been introduced relative to Udo de Haes *et al.* (1999): 'extraction' of abiotic resources is now 'depletion' of abiotic resources, the latter term being more impact-oriented; and the category 'ecotoxicity' has been broken down into five subcategories, three of which are included in Group A: freshwater aquatic, marine and terrestrial ecotoxicity.

Group B: 'Study-specific impact categories' comprises categories that may merit inclusion, depending on the Goal and scope of the LCA study and whether appropriate data are available, and for which a baseline and/or alternative characterisation method is proposed in this Guide. Most of these categories are mentioned by Udo de Haes (ed.;1996) or Udo de Haes *et al.* (1999), although some have been defined more recently (e.g. the subcategories 'freshwater sediment ecotoxicity' and 'marine sediment ecotoxicity').

Group C: 'Other impact categories' comprises the categories mentioned by Heijungs *et al.* (1992), Udo de Haes (ed. 1996) and/or Udo de Haes *et al.* (1999) for which no baseline characterisation method is proposed in this Guide. These impact categories require further elaboration before they can be used in LCA studies, with research still in progress. Desiccation, for instance, is an issue that is receiving considerable attention in the Netherlands (collaboration between KIWA and RIZA) and Australia (CRC).

[1] A new version of the EPS method has recently been published (Steen, 1999). This version could not be evaluated in the present Guide, however.

[2] Historically, the list of impact catepories has its origins in the 'environmental themes' defined in the Netherlands' first National Environmental Policy Plan (VROM, 1989); it has since been revised and refined for the LCA (and other) purposes.

[3] A 'characterisation method' for an impact category comprises a category indicator, a characterisation model and characterisation factors derived from that model.

Besides the impact categories distinguished below in Table 4.2.2, this Guide also identifies two additional rubrics: 'Interventions for which characterisation factors are lacking' and 'Economic flows not followed to system boundary', discussed in Sections 4.3.17 and 4.3.18, respectively. These two rubrics should be included in the results of every LCA study.

Table 4.2.2: Default list of impact categories and subcategories

impact category	single baseline characterisation method provided in this Guide?	other characterisation method(s) available in the Guide?
A. Baseline impact categories		
Depletion of abiotic resources	yes	yes
Impacts of land use		
land competition	yes	yes
Climate change	yes	yes
Stratospheric ozone depletion	yes	yes
Human toxicity	yes	yes
Ecotoxicity		
freshwater aquatic ecotoxicity	yes	yes
marine aquatic ecotoxicity	yes	yes
terrestrial ecotoxicity	yes	yes
Photo-oxidant formation	yes	yes
Acidification	yes	yes
Eutrophication	yes	yes
B. Study-specific impact categories		
Impacts of land use		
loss of life support functions	no	yes
loss of biodiversity	no	yes
Ecotoxicity		
freshwater sediment ecotoxicity	yes	yes
marine sediment ecotoxicity	yes	yes
Impacts of ionising radiation	yes	yes
Odour		
malodourous air	yes	no
Noise	yes	no
Waste heat	yes	no
Casualties	yes	no
C. Other impact categories		
Depletion of biotic resources	no	yes
Desiccation	no	no
Odour		
malodourous water	no	no
...	...	...

In some LCA methodologies - for example, that given in the scientific background to the EDIP method (Hauschild & Wenzel, 1998) and the Nordic report (Lindfors *et al.*, 1995c) - other default lists are used. Here, however, we proceed from the list of Table 4.2.2, which is based mainly on the work of WIA–2, because of the international support it enjoys. Currently modeled (midpoint) category indicators for the baseline impact categories recommended in this Guide with an indication of their environmental relevance are illustrated in Figure 4.2.2, freely adapted from Udo de Haes *et al.*(1999).

Although in certain contexts it may also be relevant to have information about the total energy consumed by a product system, in practice this may prove to be quite a complex issue. What precise energy requirements are to be taken, for example? The total energy used by the unit processes? The energy that can potentially be supplied by the energy resource, as originally extracted? What energy can be supplied by a particular grade of coal? If there is additional assessment of energy resources in MJ, every effort must be made to ensure that these are not subsequently double-counted in the weighting step.

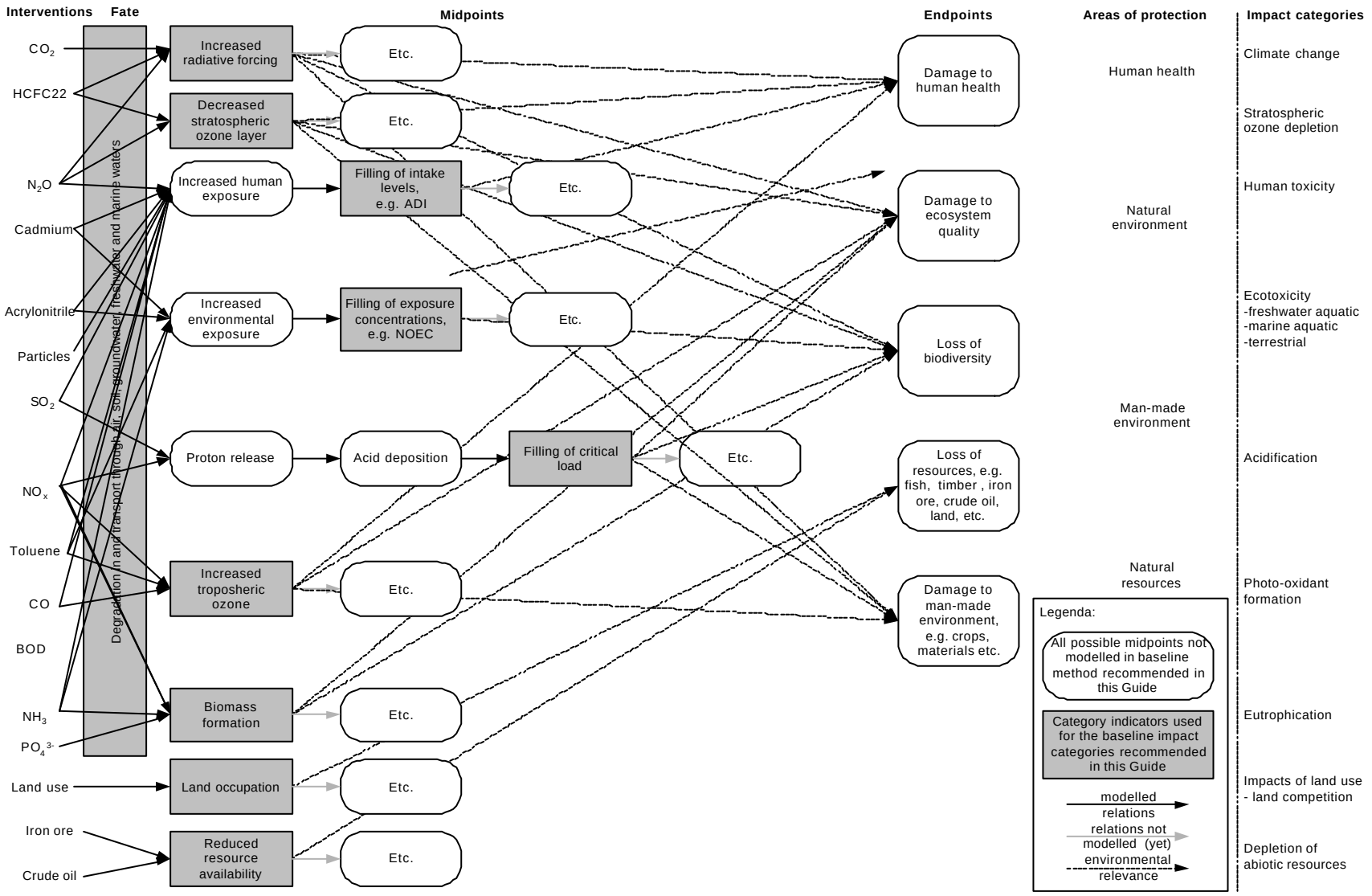


Figure 4.2.2: Currently modeled (midpoint) category indicators for the baseline impact categories recommended in this Guide with an indication of their environmental relevance (freely adapted from figure presented by Udo de Haes *et al.,* 1999).

With respect to the reporting of the various aspects of Impact assessment, the general requirements of

If a third party report according to ISO 14040 (1997E), clause 6, is prepared, the report shall include the following items:

a) the LCIA procedures, calculations, and results for the study;
b) limitations of the LCIA results relative to the defined goal and scope of the study;
c) the relationship of the LCIA results to the defined goal and scope, see annex A;
d) the relationship of the LCIA to the LCI results, see annex A;
e) impact categories considered, including rationale for their selection and a reference to their source;
f) descriptions of or reference to all characterisation models, characterisation factors and methods used, including all assumptions and limitations;
g) descriptions of or reference to all value-choices used in relation to impact categories, characterisation models, characterisation factors, normalisation, grouping, weighting, and elsewhere in the LCIA a justification for their use and their influence on the results, conclusions and recommendations;
h) a statement that the LCIA results are relative expressions and do not predict impacts on category endpoints, exceedence of thresholds, safety margins, or risks.

When included as part of the LCA study the following items shall also be included if a third party report according to ISO 14040 (1997E), clause 6, is prepared:

a) a description and justification of the definition and description of any new impact categories, category indicators or characterisation models used for the LCIA;
b) a statement and justification of any grouping of the impact categories;
c) any further procedures that transform the indicator results, and a justification of the selected references, weighting factors, etc.;
d) any analysis of the indicator results, for example sensitivity and uncertainty analysis or the use of environmental data including any implication for the results;
e) data and indicator results reached prior to any normalisation, grouping or weighting shall be made available together with the normalised, grouped or weighted results.

In addition, for comparative assertions disclosed to the public the report shall include the following items:

a) an evaluation of the completeness of the LCA;
b) a statement as to whether or not international acceptance exists for the selected category indicators and a justification for their use;
c) a justification for the scientific and technical validity and environmental relevance of the category indicators used in the study
d) the results of the uncertainty and sensitivity analysis;
e) an evaluation of the significance of the differences found;
f) if included in the LCA study:
   - the procedures and results used for grouping;
   - a statement that conclusions and recommendations derived from grouping are based on value-choices;
   - a justification of the criteria used for the normalisation and grouping (these can be personal, organisational or national value-choices);
   - a statement when grouping is used that “The ISO 14042 standard does not specify any specific methodology or support the underlying value-choices used to group the impact categories""
   - a statement when grouping is used that “The value-choices and judgements within the grouping procedures are the sole responsibilities of the commissioner of the study (e.g., government, community, organisation, etc)".

Where relevant, the items listed in this subclause should also be considered in the elaboration of other kinds of reports where LCIA results are used

Note 1: A graphical presentation of LCIA results as part of the report may be useful but one should consider the fact that this invites implicit comparisons and conclusions.
Note 2: Due to the inherent complexity of the LCIA phase, the aforementioned additional documentation beyond the requirements stated in ISO 14040 may be desirable for internal and two party reports.

ISO 14040, clause 6, apply here (see Chapter 2). In addition, ISO 14042 (2000E), clause 10, lays down the following requirements for third party study reports on Impact assessment:

Lindfors *et al.* (1995a) list the following reporting issues for Impact assessment:

– “The list of the impact categories considered in the study shall be reported. Whether the category is handled in a quantitative or qualitative way should also be noted. Deviations from the list in Table 7.1

[in Lindfors *et al.* (1995a)] should be highlighted and justified. This can be regarded as a part of the Goal definition and Scoping component [...].

- Under the heading of each impact category all inputs and outputs that cancontribute to the impact shall be noted. Inputs and outputs for which no quantitative information is available shall also be noted. If desired, as an alternative, the result of the classification can be reported together with the characterisation subcomponent.
- If a 'red-flag' classification of chemicals is performed, reference to the relevant list(s) shall be made. A report should contain a brief description of the background to the list(s), the criteria used in the list(s) and the motivation for the choice of the list(s).
- If a 'red-flag' classification of risks of accidents is performed, the criteria used for the flags shall be reported.
- [...] The reader shall be given sufficient information to be able to reproduce the results and check the data sources.
- Explanations of all methodological choices should be presented. This may be done by referring to another available document where the choices are justified.
- In cases where the results are a function of the chosen method(s), an explicit reference to the used method(s) shall be made where results and conclusions are reported.
- Not only the total contribution to each impact category should be presented, but also the contribution from each parameter considered in the classification.
- Results from the characterisation shall be presented in the form of tables and/or matrixes.
- A discussion on what differences in the results are regarded as significant should be included.
- As a part of an initial valuation, the characterisation results may be presented in a qualitative manner, using signs to indicate differences and significance of differences in tables and/or matrixes. Justifications for conclusions shall be presented where appropriate.
- If a normalisation is performed, documentation of methods and data (with adequate references), and explanations for chosen reference areas shall be included in the report.
- If valuation methods are used, the method(s) shall be described, weighting factors (including data gaps) shall be presented with references [and] arguments for the choice of method(s) should be presented. [..]"

More detailed reporting recommendations for the various impact categories are presented in Chapter 7 of Lindfors *et al.* (1995a).

In its report, the SETAC-Europe Case studies Working Group (Meier *et al.*, 1997) gives the following (minimum) reporting guidelines for the Impact assessment phase:

> "If an Impact assessment has been carried out then the methodology used should be clearly detailed in the report. The reasons for not carrying out an Impact assessment should be detailed. If the CML/SETAC approach has been used then details under the following headings should be included:
>
> *Classification*
> The selection of impact categories should reflect the goals of the study and should be justified. Any specific exclusion or inclusion of impact categories should be clearly detailed and justified.
>
> *Characterisation*
> The characterisation methodology should be detailed and explained. If any normalisation has been carried out the methodology adopted should be clearly justified including geographic and temporal considerations.
>
> *Valuation*
> If a valuation has been undertaken the methodology should be clearly explained and justified, both for quantitative and qualitative approaches."

In this Guide the name and sequence of some of the steps of Impact assessment deviate slightly from those used in the ISO standards (see Section 1.4.3), and the ISO reporting issues have therefore not been adopted precisely as they stand. However, all the ISO issues are covered in the reporting Guidelines provided in Part 2a. Furthermore, where possible and useful the ISO guidelines have been rendered more explicit, based on the guidelines provided by Lindfors *et al.* (1995a) and Meier *et al.* (1997).

PROSPECTS
No specific developments are foreseen in this area.

CONCLUSIONS
In this Guide the problem-oriented (midpoint) approach has been elaborated as a baseline for Impact assessment. The Eco-indicator 99 approach may be used as a sensitivity analysis. A default list of impact categories is furthermore proposed, distinguishing between:

Group A: Baseline impact categories	– baseline characterisation method to be selected in Section 4.3 – to be included in (almost) all LCA studies
Group B: Study-specific impact categories	– to be included if appropriate to the topic of study and if data are available – baseline and/or alternative characterisation methods available
Group C: Other impact categories	– no baseline characterisation method available – require further elaboration for inclusion in LCA

RESEARCH RECOMMENDATIONS
No specific research is recommended.

4.3 Selection of characterisation methods: category indicators, characterisation models and factors

TOPIC
The interventions recorded in the inventory table are quantified in terms of a common category indicator. To this end characterisation models are used, from which characterisation factors are derived for individual pollutants and so on. For a given impact category, a characterisation method comprises a category indicator, a characterisation model and characterisation factors derived from the model.
The impact categories distinguished in this Guide (see Table 4.2.2) are treated individually in Sections 4.3.1 to 4.3.16, below, thereby discussing the models, factors and indicators available for the category in question. Wherever feasible a baseline characterisation method is recommended which in our view represents the current best available practice.
In cases where a choice of methods was available, the arguments in favour of the preferred baseline method are presented, designating one or more of these other methods as an alternative or additional method as relevant. These choices are based on selection criteria distilled from the relevant ISO standards and the work of the second SETAC-Europe Working Group on Impact assessment, WIA–2 and these criteria will be discussed in this section.

DEVELOPMENTS IN THE LAST DECADE
With respect to the selection of characterisation methods, ISO has set out a number of starting points (see earlier textbox) for comparative assertions, as has been done more generally by SETAC WIA–1 (Udo de Haes (ed.; 1996). Combining the input from these two sources and adding further starting points related to the fact that the present Guide goes beyond the work of WIA–2, a list of relevant criteria has been drafted for the selection of the baseline characterisation methods recommended in the present Guide (see Table 4.3.1).
The criteria based on the (marginally modified) WIA–1 starting points and the additional criteria adopted in this Guide are discussed immediately after Table 4.3.1.
In the tables evaluating the baseline method recommended for each impact category (Sections 4.3.1 to 4.3.16) these selection criteria are referred to in abbreviated form, as indicated in italics in Table 4.3.1.

Table 4.3.1: Selection criteria for the baseline characterisation method recommended in this Guide, with reference to ISO and WIA–2 starting points.

	ISO starting points	WIA–2 starting points	selection criteria for baseline method recommended in this Guide
1	the category indicator should (shall for comparative assertions) be modeled in a scientifically and technically valid way in relation to the environmental interventions, i.e., using a distinct identifiable environmental mechanism and/or reproducible empirical observation	included	included
2	the category indicators and models shall be environmentally relevant, i.e. shall be sufficiently clearly related to the category endpoints, at least qualitatively	included	included
3	the category indicators and models should be internationally accepted, i.e. based on an international agreement or approved by a competent international body	not explicitly included	included
4	value-choices and assumptions should be minimised	included	included
5	category indicators can be chosen anywhere in the environmental mechanism of an impact category, from environmental interventions to category endpoints (focal point in environmental mechanism)	included	included
6	not included	it should be possible to multiply characterisation factors by mass or other units indicating the magnitude of the environmental interventions	modified: the baseline category indicators should be linear (linearity)
7.	not included	the preferred time span for fate and effects is eternity, with 100 years as a second option; all effects of the emission/ extraction occurring now and in the future should be taken into account	included
8	not included	the category indicators and models should include the modeling of fate, exposure/intake and effects, as relevant	included
9	not included	the category indicators and models should include effects below thresholds ('less is better' approach)	included
10.	not included		it should be possible to perform Impact assessment without information on time or location (time- and location-independent)
11.	not included		the method should be operational for a sufficient number of environmental interventions
12.	not included		the uncertainty margins of the indicator result should be as small as possible

Criterion 6: the baseline category indicators should be linear
In this context, linearity means that characterisation is based on characterisation factors that are independent of the magnitude of the environmental intervention. This has been adopted as a selection criterion for the baseline category indicators for two reasons:

– in this way the general structure of LCA is not altered:

$$impact_{cat} = \sum_i m_i \times characterisation\ factor_{cat,i} \qquad (4.3.1)$$

– and the quantity chosen in the definition of the functional unit, which is often arbitrary, does not influence the mutual relationship between the results for different categories.

Criterion 10: it should be possible to perform Impact assessment without information on time or location
There is broad ongoing debate on whether the location of emissions (or extractions) and receptors should be taken into account in LCIA. One of the classic examples concerns the emission of salt to the sea; the same may hold true for the SO_x and NO_x emissions of ships sailing in open sea. It would seem logical to introduce spatial information to resolve the problem of possible overestimation of impacts. If one knows the location of a given emission, one can determine whether effects are likely to occur, using information on the sensitivity of the location or the region. Similar considerations hold with respect to the fate of emissions.
There are four main approaches to location-dependent characterisation:

1. No such characterisation. LCA is regarded as a macroscopic tool of particular value for indicating the overall (potential) environmental impacts of a (product) system.
2. Distinction between sensitive and non-sensitive areas, with emissions being ignored only when non-sensitive areas are clearly involved. No fate modeling is performed.
3. Introduction of an effect-oriented site factor. The EDIP programme proposes consistent introduction of a site factor of between 1 and 0, depending on the projected sensitivity of the area in which the substance is emitted (Wenzel *et al.*, 1997). No fate modeling is performed.
4. Introduction of location-dependent characterisation factors based on fate and effect modeling. Potting *et al.* (1998) and Huijbregts (1999b) propose an approach including both fate and regional sensitivity, based on a dispersion model and a model predicting the sensitivity to acidification. For some LCA applications this may be a useful approach, but the additional inventory data required constitute a significant drawback.

These four approaches focus principally on the location of effects, with only Potting *et al.* (1998) and Huijbregts (1999b) proposing a location-dependent approach that integrates fate and effect. For a further discussion on integrated fate and effect modeling, see Section 1.2.3.3 and Wegener Sleeswijk (in prep.).

Location-dependent characterisation suffers from a number of general problems and limitations. Although methods and models often allow for location-dependent elaboration, such elaboration increases the quantity of inventory data required, making the inventory table more complex, as data on different locations cannot be aggregated. Data requirements may even become prohibitive, for example when different sites are distinguished for each impact category. Until now location-dependent factors have been proposed for different impact categories independently. A comprehensive approach distinguishing the same, limited number of regions for all impact categories would be preferable, to keep the locational differentiation in the inventory manageable.

In many LCA studies, however, no information is available on the location of emissions and/or extractions. Our baseline category indicators must therefore be able to assess at least location-independent inventory data. If, in addition, location-specific information can also be assessed with the same category indicator this is then, of course, an advantage.

Besides spatial differentiation, differentiation in the temporal domain may also be relevant for certain impact categories; see the general discussion in Section 1.2.3.3. Examples of impact categories for which temporal differentiation might be useful are day- and nighttime noise and summer and winter smog. The same reasoning can be applied to temporal as to spatial differentiation. For many interventions and unit processes there will be no temporal information available, and it should therefore at least be possible to apply non-time-specific characterisation factors.

It is not only the time of occurrence of interventions and/or impacts that may be of interest, but also their duration; see Section 1.2.2.3. In the case of the Inventory analysis, opting for a cradle-to-grave analysis already implies a basic perspective of infinite time. For impact assessment the situation is arguably different. Ecosystems are slow to adapt to changes in environmental conditions, for instance. It might therefore be argued that impacts with a very long time horizon require form of time-discounting or cut-off beyond, say, 100 or 1000 years (Udo de Haes (ed.), 1996).

Criterion 11: The method should be operational for a sufficient number of environmental interventions
A qualitatively excellent category indicator operational for only a small fraction of relevant environmental interventions is not a good choice for the baseline method, because the results obtained with the indicator should represent all the interventions contributing to the category in question.

Criterion 12 (+2): The uncertainty margins of the baseline indicator result should be as small as possible (while the indicator remains as environmentally relevant as possible)
Criterion 12 is often in conflict with criterion 2: endpoint indicators often lead to greater uncertainties in results than midpoint indicators, for example, but they have more environmental relevance. A balance should therefore be sought between these two selection criteria: indicators should be as environmentally relevant as possible without introducing undue uncertainty into the results.

Whether a method is based on a proportional or a marginal approach plays no role in the choice of baseline. As current characterisation models employ either linear relationships (e.g. for toxicity, acidification) or non-linear relationships, it is not possible to derive both proportional and marginal characterisation factors. While such a theoretical distinction can be made, then, this choice rarely presents itself in practice and we shall have to work with the (heterogeneous) factors we have. (See Section 1.2.3.4 for a more extensive discussion of this and related topics.)

PROSPECTS
The new SETAC-Europe Working Group on life cycle Impact assessment (WIA–2) and the forthcoming SETAC/UNEP cooperation have been charged with preparing an authorised, recommended list of impact categories with respective category indicators and characterisation factors. These may differ from the categories, indicators and factors adopted here, in which case these documents will need to be updated in due course (in about 3–4 years).

CONCLUSIONS
On the basis of the selection criteria presenter above, in this Guide a distinction has been made between:
- a baseline characterisation method, i.e. the method recommended here as the current best available practice for the impact category in question;
- alternative characterisation methods, which may be adopted instead of the baseline method if duly justified and documented, or may be used in tandem with the baseline method, as a sensitivity analysis;
- additional characterisation methods, which may be applied similarly to alternative methods, but requiring additional effort (e.g. collection of additional data, development of additional models);
- variant characterisation methods, which start from entirely different principles.

RESEARCH RECOMMENDATIONS
Short-term research
- It is recommended to investigate the usefulness and feasibility of category indicators for the impact categories ‘human toxicological impacts’ and ‘ecotoxicological impacts’ calculated using the damage approach.
- The marginal and proportional category indicators currently used in LCIA should be inventoried.
- A comprehensive site- or location-dependent approach should be developed (although the relevant level of differentiation might vary among impact categories).
- Appropriate time horizons for LCIA should be examined in more detail.

Long-term research

- It is recommended to investigate the potential for developing category indicators for impact categories within an overall framework based on a damage approach. As a start, the scope could be investigated for developing category indicators for human and ecosystem health, including not only toxicological impacts but also such other impacts as casualties, smog, etc.
- It is recommended to study, for several impact categories, the differences between LCIA results based on proportional and a marginal modeling and the influence of including/excluding background concentrations.
- The scope for fully integrating fate in the characterisation factor for each impact category should be investigated.
- Because of the wide variation in the dilution volume of substances, the scope for including this parameter in multimedia models should be investigated.

In the following sections each of the impact categories listed in Table 4.2.2 is described and documented and a baseline characterisation method recommended.

4.3.1 Depletion of abiotic resources

TOPIC

'Abiotic resources' are natural resources (including energy resources) such as iron ore, crude oil and wind energy which are regarded as non-living. Abiotic resource depletion is one of the most frequently discussed impact categories and there is consequently a wide variety of methods available for characterising contributions to this category. To a large extent these different methodologies reflect differences in problem definition. Depending on the definition, this impact category has only natural resources, or natural resources, human health and the natural environment as areas of protection (see Figure 4.2.2).

DEVELOPMENTS IN THE LAST DECADE

Three types of abiotic resources can be distinguished: deposits, funds and flows. Deposits are resources that are not regenerated within human lifetimes. Examples of deposits are fossil fuels, minerals, sediments, clay, etc. Funds are resources that can be regenerated within human lifetimes. Groundwater and soil are examples of funds. Flows are resources that are constantly regenerated, such as wind, river water and solar energy (Finnveden, 1996a). It is debatable whether all three types of abiotic resources can or should be aggregated into one measure for abiotic depletion. It will be difficult to combine Impact assessment for flows, for which there is no reserve to be depleted but rather a maximum utilisable flow, with that for deposits and funds (Guinée & Heijungs, 1995).

Heijungs *et al.* (1992)

In Heijungs *et al.* (1992) for a given resource i, abiotic depletion was defined as the ratio between the quantity of resource extracted (m_i) and the recoverable reserves of that resource (M_i):

$$Abiotic\ depletion = \sum_i \frac{m_i}{M_i} \qquad (4.3.1.1)$$

yielding a dimensionless indicator result. The units used for both extractions and reserves could thus be freely selected, as long as this was consistent for a given resource. Ores were normally expressed in kg and natural gas in m^3, although MJ could be used as an alternative. "Abiotic depletion" also covered depletion of some energy resources such as fossil fuels.

Heijungs *et al.* (1992) observed that this is a simplified method and that it should ultimately be extended to include the extraction rate, expressed in kg/yr or m^3/yr.

Heijungs *et al.* (1997) make a distinction between resources that can be depleted and those that are competitively used, stating that deposits fall into the former category and flows into the latter, while funds may fall into either category. They state that the two categories should be assessed using two different methods: resources that are depleted should be assessed by a method based on depletion, those that are competitively used by a method based on competition. One implication of this is that aggregation of abiotic resources into a single measure is not meaningful.

Reviews of existing Impact assessment methods for the depletion of abiotic resources are provided in several publications, in particular Heijungs *et al.* (1992), Fava *et al.* (1993), Guinée & Heijungs (1995), Lindfors *et al.* (1995a,c), Lindfors (1996), Finnveden (1996a) and Heijungs *et al.* (1997). Broadly speaking, the available methods fall into six groups, reviewed very briefly below (mainly Finnveden, 1996a, and Lindfors *et al.*, 1995c, supplemented with more recently developed methods).

1. No assessment or aggregation (e.g Lindfors, 1996).
2. Aggregation of natural resource extractions on a mass basis (e.g. Lindfors *et al.*, 1995c).
3. Aggregation and assessment based either on a) 'ultimate reserves', i.e. the quantity of resource (as a chemical element or compound) that is ultimately available, estimated by multiplying the average natural concentration of the resource in the primary extraction media (e.g. the earth's crust) by the mass or volume of these media (e.g. the mass of the crust) (Guinée, 1995); or b) on 'economic reserves', i.e. that part of the reserve base[1] which can be economically extracted at the time of determination (United States Department of the Interior - Bureau of Mines, 1993) and/or current extraction rate. The method of Heijungs *et al.* (1992) is an example of assessment based on reserves. Alternative assessment methods proceed from resource extraction rates relative to reserves (see Guinée & Heijungs, 1995 and Ekvall *et al.*, 1997), from extraction rates only (Goedkoop, 1995) or from per capita reserves (see Hauschild & Wenzel, 1998)
4. Aggregation and assessment based on the cost of 'restoring' the resource to its original, natural state, or on the costs associated with substituting current extraction processes by presumed 'sustainable' processes. Pedersen (1991) and Steen (1995) describe such methods.
5. Aggregation and assessment based on energy content or exergy content or consumption (e.g. Finnveden, 1996b; see also Ayres *et al.*, 1996 and Ayres, 1998). Exergy is the amount of energy that can be obtained when matter is brought reversibly into equilibrium with its surroundings. It is the fraction of the energy content that can be used for work (= available energy). The exergy consumed is the exergy of the resources extracted as input minus the exergy of the outputs (Finnveden, 1996b). Finnveden suggests that the potential exergy of an ore might be used as a measure for depletion of abiotic resources in LCA.
6. Aggregation and assessment based on the change in the anticipated environmental impact of the resource extraction process due to lower-grade deposits having to be mined in the future. This method is described by Blonk *et al.* (1997a) and Müller-Wenk (1998) among others. It has been operationalised for metal ores and energy resources. In the case of metal ores, the virtual additional energy that will be required for future extraction processes is estimated. This additional energy, and the energy content of energy resources, are converted into the weighting indices of the Eco-indicator 95 approach by performing an LCA for the transformation of 1 kg 'heavy-grade' oil into thermal energy. Within the Eco-indicator 99 methodology a method has been developed based on that of Müller-Wenk (1998) (Goedkoop & Spriensma, 1999).

The authors differ in their conclusions as to the best method for characterising abiotic resource depletion. Lindfors (1996), for instance, recommends not aggregating abiotic recources at all in the characterisation phase of LCAs conducted under the ecolabeling programme (= method group 1, above). He states that all characterisation methods introduce value-based judgement and that there is no consensus on this point. Finnveden (1996a) states that further discussion is necessary before one particular methods can be selected. A major debating point will be problem definition. In our view, however, the current lack of any aggregation of abiotic resources at all has the disadvantage that the outcome of the characterisation phase comprises many separate scores for this impact category, which the LCA practitioner will probably not put to any use. The implication is that the problem is thus neglected. Hauschild & Wenzel (1998) also advise against applying aggregation in the characterisation phase, proposing use of weighting at a later stage, after normalisation. In their weighting method resource depletion is specified as a fraction of known per capita economic reserves in 1990. They eventually therefore recommend using a method from group 3, above. The only difference from other methods in group 3 is that they advise aggregating in a later phase of the LCA, during weighting rather than characterisation. In their review, Heijungs *et al.* (1997) employ five criteria to assess all the methods then available. They conclude that there is no 'perfect' method meeting all their criteria.

The main conclusion here is that there is as yet no consensus about what constitutes the best category indicator for 'abiotic depletion', the choice depending crucially on the definition of this term. Although in

[1] The reserve base is that part of an identified resource that meets specified minimum physical and chemical criteria related to current mining practice (United States Department of the Interior - Bureau of Mines, 1993).

most publications the problem of abiotic depletion is not precisely defined, four groups of definitions can be broadly distinguished, based on what is seen as the key problem:

A. the decrease of the resource itself – method groups 2 and 3
B. the decreasing reserves of useful energy/exergy in the world – method group 5
C. the contribution of current extraction processes, or possible 'restoration' of the resource, to other impact categories (in the first case, this means that resource depletion is not in fact regarded as a separate environmental problem at all, as the environmental impacts of extraction processes are already included in LCAs (*cf.* Finnveden, 1996a) – method groups 1 and 4
D. the change in the environmental impact of extraction processes at some point in the future (e.g. as a result of having to extract lower-grade ores or recover materials from scrap) – method groups 4 and 6.

Depending on the problem definition, this impact category is associated with various areas of protection: natural resources (option A and B), human health and the natural and man-made environment, now (option C), or natural resources, human health and the natural and man-made environment, in the future (option D).

In order to select a baseline method it is first necessary to opt for one of the available definitions. Option D is not consistent with the methods adopted for the other impact categories, where the concern is consistently not with the impact of future changes in processes and interventions, but with those of current interventions. Future changes in processes and interventions constitute changes in the product system and should be accounted for in the Goal and scope and Inventory phases, not during Impact assessment for a particular impact category. Under option D all impacts would be based on the changed product system, and not only abiotic depletion.

Option C means that abiotic depletion is not deemed a relevant impact category. No separate assessment is then necessary, for if the Inventory analysis is correctly performed all environmental impacts should already be included within other impact categories.

Thus, options C and D are rejected, leaving options A and B.

If the decrease of the resource itself is taken as the key problem (option A), assessment based on reserves and/or current extraction rates appears to be the best available method (method group 3). This method relates resource extractions obtained from the Inventory analysis directly to existing reserves and/or (annual) extraction rates. There is still room for debate on several points, however:

– Which of the many types of reserves do we consider: economic reserves, ultimate reserves in primary media (ore, fossil fuels) or also reserves in the economy (in products, scrap)?
– Should we focus on reserves or on extraction rate, or should both be included?
– Should we take into account the economic value of the resource?

According to Guinée & Heijungs (1995) a method based on ultimate reserves and rates of extraction is the best option, as these parameters best indicate the seriousness of resource depletion. As the notion of 'economic reserves' involves a variety of economic considerations not directly related to the environmental problem of resource depletion, 'ultimate reserves' appears to be a more appropriate yardstick. In their proposed method the indicator result is expressed in kg of a reference resource (antimony):

$$abiotic\,depletion = \sum_i ADP_i \times m_i \qquad (4.3.1.2)$$

with:

$$ADP_i = \frac{DR_i}{(R_i)^2} \times \frac{(R_{ref})^2}{DR_{ref}} \qquad (4.3.1.3)$$

and:

ADP_i	Abiotic Depletion Potential of resource i (generally dimensionless);
m_i	quantity of resource i extracted (kg);
R_i	ultimate reserve of resource I (kg);
DR_i	extraction rate of resource i ($kg \cdot yr^{-1}$)
R_{ref}	ultimate reserve of the reference resource, antimony (kg)
DR_{ref}	extraction rate of R_{ref} ($kg \cdot yr^{-1}$)

The indicator result is expressed in kg of the reference resource, *viz.* antimony.

This method is partly operational. Guinée (1995) has developed ADPs for many elements, using antimony as the reference element. Thus far, only ultimate reserves have been included. However, these ADPs for elements need to be updated and converted to ADPs for resources, as it is these that are documented in the Inventory analysis. These resources are mainly compositions of several elements. The method is suitable for depletion, but not for competitive use, i.e. use of a resource that restricts the potential for others to use that same resource.
If the scarcity of useful energy/exergy in the world is taken as the key problem (option B), it should be borne in mind that the resources will be 'valued' on the basis of exergy content only. However, it is debatable whether the value of the metals used in a particular process is dependent on the exergy content of the ore in question.

Separate indicators for energy resources
As discussed in Section 4.2, it may sometimes be relevant to *add* information about the total amount of (fossil) energy consumed by a system. One common approach is then simply to aggregate all forms of energy consumption, by multiplying the energy resources extracted from the environment by their respective (gross) heating value, this being the best indication of the extent to which the energy reserve is actually depleted (see Frischknecht *et al.*, 1998). Frischknecht *et al.* (1998) propose including not only fossil fuels but also biofuels (wood, etc.), solar and wind energy, etc. as well as nuclear resources (uranium, etc.). These energy resources obviously cannot be aggregated on the basis of heating value alone, and this is therefore not recommended as an additional method in this Guide. If practitioners nonetheless opt to undertake additional assessment of energy resources in MJ, due care should be taken to avoid double-counting in the weighting step.

PROSPECTS
The subject of abiotic resource depletion is still being widely debated and much research is still in progress on developing indicators for this impact category, especially within method groups 5 and 6, above.

CONCLUSIONS
In this Guide we consider resource depletion to be an environmental problem in its own right, while recognising that views differ as to the precise definition of the resource problem. As argued, though, we consider that ultimate reserves and extraction rates together best reflect the seriousness of resource depletion (see Guinée & Heijungs, 1995). We therefore recommend as a baseline the method employing these parameters developed by Guinée & Heijungs (1995; see also Guinée, 1995). As mentioned above, there is still a need to convert the ADPs for elements to ADPs for composite resources. It should also be borne in mind that this method covers depletion but not competitive use. Table 4.3.1.1 indicates how this method scores with regard to the (ISO-based) criteria of Table 4.3.1.

Table 4.3.1.1: Evaluation of the baseline characterisation method for abiotic depletion, using the characterisation factor ADP, with respect to the (ISO-based) criteria of Table 4.3.1.

	criterion	evaluation
1.	scientifically and technically valid	not relevant; no environmental mechanism involved (actual resource extraction is assessed)
2.	environmentally relevant	yes, if problem definition A is adopted
3.	internationally accepted	no; method not authorised by an international body
4.	value-choices and assumptions	yes; problem definition implies a value-choice
5.	focal point in environmental mechanism	not relevant; no environmental mechanism involved
6.	linearity	yes
7.	time span	not relevant
8.	fate, exposure/intake and effects	not relevant
9.	less is better	yes, no threshold

	criterion	evaluation
10.	time- and location-independent	yes
11.	operational	partly; factors still to be worked up from elements to resources
12.	uncertainty margins	considerable uncertainty about magnitude of current reserves

Three alternative methods are included in this Guide as options for sensitivity analysis, viz. methods using:

- a characterisation factor based on extraction rates and *economic* (rather than *ultimate*) reserves, i.e. that part of the reserve base which can be economically extracted at the time of calculation;
- a characterisation factor based only on ultimate or economic reserves (R) and not on extraction rates: characterisation factor = 1/R;
- the factors based on exergy content developed by Finnveden (1996b) or Ayres *et al*. (1996), useful when focusing on declining global energy/exergy content.

Recommendation for extended LCAs:

- recalculate ADPs for minerals (i.e. compositions) rather than chemical elements.

method status	characterisation method/factor	reference
baseline	based on ultimate reserves and extraction rates	Guinée & Heijungs, 1995
alternative 1	based on economic reserves and extraction rates	Guinée & Heijungs, 1995, adapted
alternative 2	based on ultimate or economic reserves only	Guinée & Heijungs, 1995, adapted
alternative 3	based on exergy content	Finnveden, 1996b; Ayres *et al*., 1996
additional	–	–
variant	–	–

Additional remark

Because of the difference between flows on the one hand and deposits and funds on the other, it is not possible to aggregate all abiotic resources into one measure using any of the methods currently available. Until such time as a method for competitive use has been developed, it will not be possible to incorporate flows in the method.

RESEARCH RECOMMENDATIONS

Short-term research:

- To develop up-to-date ADPs research should be undertaken to establish contemporary data on reserves of abiotic resources and extraction rates. Using these data the ADPs for chemical elements of Guinée (1995) should be updated and extrapolated to yield ADPs for minerals and composite ores (see also: Recommendations for extended LCAs).
- It should be investigated whether abiotic depletion can be split into two subcategories: depletion of energy resources and of other resources, operationalising a dual methodology and subsequently reaggregating the results (among other requirements, the same units must then of course be used).

Long-term research:

- Continued debate is required on defining the problem of abiotic depletion in LCA. Once a conclusion has been reached, the best available method based on the chosen definition should be further developed, for most methods are as yet only partly operational.
- It might be useful to develop a method covering competitive use of flows (and funds) and to investigate the scope for aggregating flows, deposits and funds. The scope for distinguishing between competitive use and depletion in LCA should also be investigated. Some of the methods described above have potential for including competitive use, especially for minerals that accumulate in the economy (methods based on reserves, for instance). Methods based on total availability of

reserves, including those in the economy, cover competitive use rather than depletion. Methods based on changes in future environmental impact might also be applied to competitive use if they were amended to incorporate extraction from scrap, etc., in addition to extraction from lower-grade ore.

4.3.2 Depletion of biotic resources

TOPIC

'Biotic resources' are material resources (including energy resources) regarded as living, e.g. rainforests, elephants. Depending on the precise definition adopted, this impact category has only natural resources, or natural resources, human health and the natural and the man-made environment as areas of protection (see Figure 4.2.2).

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)

The method used by Heijungs *et al.* (1992) to characterise biotic resources considers reserves and deaccumulation rates (see below):

$$Biotic\,depletion = \sum_i BDF_i \times m_i \qquad (4.3.2.1)$$

$$BDF_i = \frac{DR_i}{(R_i)^2} \qquad (4.3.2.2)$$

The indicator result is expressed in yr^{-1}. BDFi ($kg^{-1}{\cdot}yr^{-1}$, $number^{-1}{\cdot}yr^{-1}$ or $m^{-3}{\cdot}yr^{-1}$) is the Biotic Depletion Factor of resource *i*, m_i (kg) is the quantity of resource *i* extracted, R_i (kg, number or m^3) is the reserve of resource *i* and DR_i ($kg{\cdot}yr^{-1}$) is the deaccumulation rate of resource *i*, which is defined as the extraction rate, expressed in kg/yr, number/yr or m^3/yr, minus the regeneration rate, expressed in the same units.

Most developments since 1992 in the area of resource depletion have already been discussed in Section 4.3.1. Most of these tend to focus on abiotic resources. Heijungs *et al.* (1997) provide an overview focused particularly on biotic resources. Based on these reviews, supplemented with several methods developed since, three groups of methods can be distinguished for characterising biotic resources:

1. no aggregation (Lindfors, 1996)
2. aggregation based on reserves and deaccumulation rates (see Heijungs *et al*. (1992) as described in the textbox) and aggregation based on (regional) production rates divided by (regional) regeneration rates.
3. methods that also cover impacts on biodiversity and life support functions. This approach is explored by Sas *et al.* (1997), who propose three types of indicator results:

$$S_1 = \sum_i m_i \times \frac{Trepro_i}{R_i} \qquad (4.3.2.3)$$

to indicate the risk of species extinction by harvesting an individual;

$$S_2 = \sum_i A_i \times Trecov_i \times r_i \qquad (4.3.2.4)$$

to indicate the risk of species extinction by disrupting an ecosystem; and

$$S_3 = \sum_i A_i \times Trecov_i \times NPP_i \qquad (4.3.2.5)$$

to indicate the decrease in life support functions, i.e. ecological structures and processes that sustain the productivity, adaptability and capacity for renewal of lands, water and/or the biosphere as a whole (after IUCN/WWF/UNEP, 1991).

In these equations m_i is the quantity of biomass of species i extracted (in kg fresh weight or units), $Trepro_i$ the reproduction (regeneration) time of species i (yr), $Trecov_i$ the recovery time for species density or primary production in ecosystem i (yr), ρ_i the species density within ecosystem i (number

of species per standard unit of area), A_i the area of ecosystem i disrupted and NPP_i the net primary production of that ecosystem.

According to Heijungs *et al.* (1997) the method of Heijungs *et al.* (1992), based on reserves and deaccumulation rates (= method group 2) still appears to the most practicable, while the method of Sas *et al.* (1997; method group 3) is the most promising, although not yet operational. However, the last two formulae proposed by Sas *et al.* (1997) belong more to the impact category Impacts of land use', subcategory 'Loss of biodiversity and life support functions' employed in this Guide (Section 4.3.3.2). They are very similar to the formulae developed for land use by Lindeijer *et al.* (1998). The first formula of Sas *et al.* (1997) in fact represents a method based on reserves, boiling down to assessment based on the inverse of maximum regeneration, viz. m_i divided by $regeneration_i$, the latter in $kg \cdot yr^{-1}$.

An entirely different approach would be to collect four types of inventory data:
- timber with a Forest Stewardship Council (FSC) certificate (in kg);
- timber without an FSC certificate (in kg);
- fish with a certificate for marine stewardship (in kg); and
- fish without a certificate for marine stewardship (in kg).

These four groups could than be assessed and aggregated into a single category indicator providing an overall indication of the sustainability of harvest.

Concluding, once again a choice of 'best available procedure' depends on the definitions adopted. In the case of biotic depletion there are, broadly speaking, two groups of definitions[1]:
A. definitions focusing on the decrease of the resource itself as the main problem – method group 2
B. definitions focusing on the environmental impacts of resource extraction processes – method groups 1 and 3.

Depending on the definition adopted, this impact category has natural resources (option A) or natural resources, human health and the natural and man-made environment (option B) as areas of protection.

Option B would mean that all relevant impacts are accounted for under other impact categories (land use, for example), leaving no separate impact category 'biotic depletion'. We therefore focus on the decrease of the resource itself as the main problem (option A). Assessment based on reserves and/or current deaccumulation rates appears to be the best available method for this problem definition (method group 2). This method relates resource extraction directly to current reserves and/or depletion rate. There still remain several debatable issues, however:
- Which reserves are to be considered: only those 'in the wild' or also reserves in zoos?
- Should the focus be solely on reserves or also on deaccumulation rate, i.e. including regeneration? Because biotic resources generally behave more like funds than deposits, the regeneration part of the deaccumulation rate is here a particularly important topic to consider.
- Should we take into account the economic value or intrinsic value of the resource?

The first formula of Sas *et al.* (1997) does not include deaccumulation rate but only maximum annual regeneration. In the view of Guinée & Heijungs (1995) the following method is the best within this group, because it considers both reserves and rates of deaccumulation:

$$abiotic\,depeletion = \sum_i BDP_i \times m_i \qquad (4.3.2.6)$$

$$BDP_i = \frac{DR_i}{(R_i)^2} \times \frac{(R_{ref})^2}{DR_{ref}} \qquad (4.3.2.7)$$

with:

BDP_i	Biotic Depletion Potential of resource i, (generally dimensionless);
m_i	quantity of resource i extracted (kg or number)
R_i	reserve of resource i (kg or number)
DR_i	deaccumulation rate of resource i ($kg \cdot yr^{-1}$ or $number \cdot yr^{-1}$)
R_{ref}	reserve of African elephants, the reference resource (kg or number)
DR_{ref}	deaccumulation rate of R_{ref} ($kg \cdot yr^{-1}$ or $number \cdot yr^{-1}$).

[1] The other two types of definitions discussed above for abiotic depletion are not considered for biotic depletion.

The indicator result is expressed in numbers of the reference resource, e.g. the African elephant.

PROSPECTS

The subject of biotic resource depletion is still being widely debated and there is sure to be additional research on developing indicators for this impact category.

CONCLUSIONS

Neither of the aforementioned methods has yet been operationalised for more than a handful of species. Guinée (1995) has developed BDPs for a few biotic resources, using the African Elephant as a reference resource. Sas *et al.* (1997) have developed factors for timber and fish only (selection criterion 12, Table 4.3.1). No baseline method can therefore be recommended for this impact category. For extended LCAs in which biotic depletion is anticipated to play a significant role, it may be useful to calculate characterisation factors according to the method of Guinée (1995).

This Guide recommends no particular methods for sensitivity analysis.

Recommendations for extended LCAs:

- In extended LCAs in which biotic depletion is likely to be relatively prominent, characterisation factors may be calculated according to Guinée (1995).
- A category indicator based on the certification of timber and fish might be developed.

method status	characterisation method/actor	reference
baseline	–	–
alternative	–	–
additional	based on reserves and deaccumulation rate	Guinée, 1995
variant	–	–

RESEARCH RECOMMENDATIONS

Short-term research:

- BDPs based on reserves and/or deaccumulation rates should be developed. To develop these factors, research should be undertaken to establish the most recent values for reserves, extraction rates and regeneration rates.

4.3.3 Impacts of land use

The category 'Impacts of land use' covers a range of consequences of human land use. It is a relatively new topic in LCIA and still being debated and developed. In the cited SETAC-Europe WIA–1 report (Udo de Haes,1996) a distinction was made between use of land with impacts on the resource aspect and use of land with impacts on biodiversity, life support functions, etc. On the intervention side a distinction is often made between land occupation[1] (i.e. occupancy and use) and land transformation (i.e. changing its quality). On the impact side Guinée & Heijungs (1997) distinguish between competition (reducing the total stock of available land regardless of its quality) and depletion (exhausting the total reserve of a specific class of land quality). Occupation is then associated with competition, and transformation with depletion. These different aspects of intervention and impact have not yet been integrated into a single conceptual scheme for land use for use in Impact Assessment. This topic is presently being addressed by SETAC WIA–2, in the subgroup on land use impacts (Lindeijer, 2000).

We first describe the conceptual framework now under discussion in SETAC WIA–2. Given its developmental status, there are still several key areas of debate, some of which will be addressed towards the end of this section. Despite its open-ended nature and potential shortcomings, the conceptual scheme of SETAC WIA–2 has here been adopted for the principal reason that the majority of experts working in the field of land use impacts participate in this forum. The operational methods for

[1] The term 'occupation' is intended solely in the neutral sense of 'occupancy', with no political implications, to mean human use of a particular area of land for a given duration.

assessing land use impacts described in Sections 4.3.3.1 and 4.3.3.2 will be evaluated within this SETAC WIA–2 framework.

SETAC WIA–2 framework for assessment of land use impacts

On the intervention side the SETAC WIA–2 framework distinguishes two aspects of land use: occupation and transformation.

Occupation versus transformation

Land use is a direct physical intervention in the environment, attributable to a unit process. Two aspects can be distinguished:

- the associated changes in the *quality* of the land: *transformation*, typically expressed in terms of biodiversity and/or life support functions; and
- the *length of time* for which the land is used: *occupation*.

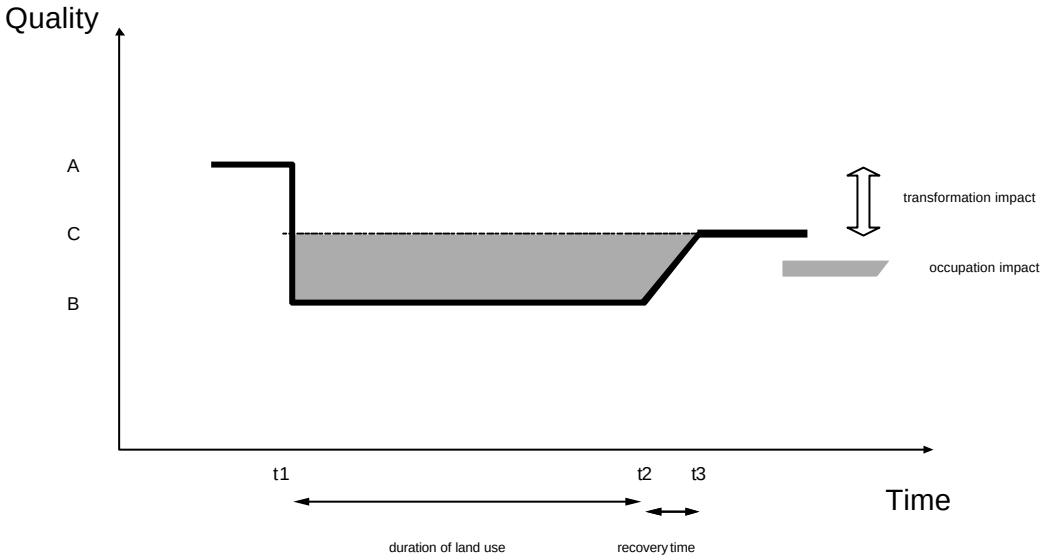


Figure 4.3.3.1: Quality of an area of land before, during and after human use between time t_1 and t_2 (e.g. expressed in terms of biodiversity and/or life support functions).

Figure 4.3.3.1 illustrates these two aspects of land use.

1. Transformation

   Land transformation is the process of changing aspects of biodiversity and life support functions, e.g. the flora, fauna, soil or soil surface from its initial state to an altered state. The altered state (level B in Figure 4.3.3.1, which may represent a state of lower or higher quality than the initial level A) may be temporary. After the termination of human activity (at t_2) the flora, fauna, soil or soil may undergo a certain degree of recovery (with or without human intervention), eventually attaining a new steady state: level C in Figure 4.3.3.1, which may represent a state of lower, higher or identical quality than or to the original level A[1]. The difference between level A and level C is the net impact of transformation. *The net transformation impact represents (the effects of) the permanent or irreversible changes in the quality of an area of land.* The transformation impact is expressed in units of quality*area The unit of this aspect of the intervention is thus square metres only.

2. Occupation

   Occupation refers to the time period during which the land is unavailable for other uses, i.e. the duration of the change of quality, how long the altered state is maintained. It thus includes both the duration of human land use and also the time taken for a new steady state to be reached,

[1] If there is no such recovery, B is the final state in Figure 4.3.3.1.

i.e. the recovery time[1]. In Figure 4.3.3.1 the occupation impact is shown as a shaded area between the curve and the final level C against which the change is measured. *The occupation impact represents (the effects of) the temporary changes in the quality of an area of land.* The occupation impact can be expressed in units of quality*area*time. This aspect of the intervention can be expressed in terms of quality*area*time and the unit is therefore square (kilo)metres*years.

Points of discussion

Figure 4.3.3.1, above, does not distinguish between the interventions and impacts associated with human land use, while such a distinction is necessary for the purposes both of discussion and of actual Inventory analysis and Impact assessment.

From the perspective of interventions, four aspects should be distinguished instead of two: - a change of state due to the activity (transformation)

- a state during the activity (occupation);
- the area required by the activity;
- the time required by the activity.

Observe that the time and area aspects can be expressed as a cardinal variable (100 days, 5 km^2), whereas the (change of) state requires nominal variables (from state A to state B). As is the case with noise, time and area aspects can be combined into a single parameter, area×time (e.g. 500 km^2×days). The change of state can also be combined with the area (5 km^2 from state A to state B). Thus, it is possible to describe a unit process with reference to two intervention items:

- change in state×area (transformation);
- state×area×time (occupation).

In a discussion of the inventory-related issues, 'state' is perhaps a better concept than 'quality'. Note that the term ‘state’ is not a state in terms of state indicators (e.g. concentrations, plant density, etc.). In this context, the term ‘state’ stands merely for the type of land use, a nominal term with no quality aspects, deliberately omitted from the present inventory-based exposition because assessment of quality is an aspect of impact assessment. Types of land use include ‘natural forest’, ‘silvicultural plantation’, ‘pasture’, ‘arable land’, ‘roadway’ and ‘built-up area’.

Choice of the final state for measuring transformation and occupation impacts

Above, the transformation impact was defined as the difference between the initial and the final steady state (level C in Figure 4.3.3.1). Lindeijer (2000.) state that the (temporary) occupation impact should therefore be defined in such a way as to avoid any overlap with the irreversible transformation impact, while covering all impacts not captured in the transformation impact. According to Lindeijer (2000) this can only be done by relating the current state to the final steady state: level C in Figure 4.3.3.1, representing the final steady state to which the land would recover (with or without human aid) if occupation were to end immediately: the current recovery potential, Thus, the occupation impact is measured as the product of the duration of the change (incl. recovery time) and the current recovery potential. times the area. As the SETAC WIA–2 workgroup states, data availability may prove to be a limiting factor for using a final steady state as a reference, however.

Points of discussion

In Figure 4.3.3.1 the assumption that human activity is of finite duration is an essential prerequisite for modeling the impacts of land use. Both the transformation and the occupation impact are defined with reference to a final steady state, the recovery potential or relaxation potential. As stated by the SETAC WIA–2 workgroup, data availability and data uncertainty may prove to be a limiting factor for using a final steady state as a reference, however. The method assumes that, for any type of land use in any kind of initial state, a final state can be estimated to which the particular area of land will recover after termination of the activity in question. If not in principle impossible, the choice of a certain final state will at least lead to speculations and/or major uncertainties.

[1] A human activity of no duration, i.e. involving only a transformation at time t1, thus still has both transformation and occupation impacts. A human activity with a certain duration, and which changes the current state but does not affect the level of the final steady state, has no transformation impact but only an occupation impact.

Allocating land transformation

Attributing land transformation to functional units is problematical, as the relationship between the two is generally anything but transparent. What will be the agricultural output from an area of cleared forest, and what proportion of the clearing is to be attributed to one kilogram of any one of these crops? How many cars and trucks will use a new rural road and what share of the associated land transformation should be allocated to a functional unit of traffic or transport? One approach to this problem is as follows

> Attribution of land transformation and occupation to a functional unit: an example from SETAC WIA–2
> Assume that a plot of land measuring 100 m^2 is utilised for a period of 10 years, thereby delivering 5 functional units.
> For 1 functional unit the intervention associated with transformation is then 20 m^2 (=100 m^2 / 5 f.u.) and that associated with occupation 200 m^2.yr (= 100 m^2 * 10 years) / 5 f.u.).
> The occupation can also be calculated using the annual figures. Use of 100 m^2 for 1 year (i.e. an occupation of 100 m2.yr) delivers 0.5 f.u. So for 1 functional unit the occupation is 200 m^2.yr (= 100m^2.yr .yr^{-1}/ 0.5 f.u.yr^{-1})

In many cases an activity performed in the service of a given functional unit is not be accompanied by any significant degree of land transformation, merely occupying land, as in the case of road traffic, sustainable silviculture and agriculture. At the macro level, however, the area of land made available for that activity in a particular country may change over time, according to policy or private initiative, and there will then be a quantitative increase or decrease in the man-made resource 'land type X'. The transformation associated with this change in land use can then be attributed to a trend (rise or decline) in the economic output of 'land type X' over a particular number of years.

In the Netherlands, for example, over a particular 10-year period $8x10^7$ m^2 of agricultural land was converted into (9500 km of) roadway, while in the same period domestic road transport grew by $16x10^9$ car-kilometres and $5.2x10^9$ tonne-kilometres. In such cases where reliable statistics are available, land transformation can thus be allocated to a functional unit of traffic/transport in the same way as land occupation. For activities of finite duration such as mining, the activities associated with both transformation and occupation can be more readily attributed to a functional unit, as the aggregate output over this period is generally known or at least estimable.

An increase in a certain land type area can also be seen as a core process to be analysed (building a road on agricultural land, converting tropical forest into a silvicultural plantation, or either of the latter into agricultural land). This process includes land transformation as one of the main impacts and transportation capacity as the main economic outflow. Whether this local transformation is part of a generic trend may not be immediately obvious, but regarding it as such may be the only way to solve the allocation problem within the framework of LCA, especially when the future economic output of the activity in question cannot be established with any certainty. The other approach to the problem of local land transformation is to abandon the life cycle perspective altogether and mark land transformation as a major environmental aspect that is simply not compatible with the principles of LCA. This approach, which implies 'forgetting' that the service-delivering activity (involving transformation, e.g. the forest or the road) is not only *produced* but also *used* for an indefinite amount of time, may be adequate for an environmental impact assessment, EIA, of the activity (rather than an LCA). From a life cycle perspective, however, attributing all local transformation to that particular activity is an overestimate as it ignores the temporal aspect of the 'use phase'.

Points of discussion

As mentioned earlier, an essential assumption of the model used for assessing the impact of land use (Figure 4.3.3.1) is that the human activity in question is of finite duration, implying that the duration of each activity can be estimated beforehand. If not in principle impossible, this will at least lead to speculations and/or major uncertainties. In the example of allocating land transformation to vehicle-kilometres: on relatively quiet roads, too, traffic volume has risen over the past 10 years, so national trends are not a good measure. Particular roads may even have been closed, although total vehicle-kilometres have risen.

Allocating land occupation

Land occupation must also be allocated to the functional unit. This is probably less problematical then in the case of land transformation, however, as annual occupation [m^2.yr] and annual economic output are

generally both known quantities. Thus, occupation can readily be attributed to economic output.

Reversibility of land transformation
A related problem with respect to land transformation is its reversibility. In the above approach to allocating transformation to land-occupying activities, the past trend is extrapolated, yielding either a net positive or net negative transformation. Future natural recovery and/or human rehabilitation efforts are thereby incorporated only in as far as proven in the past and then converted to an average trend. If, on the other hand, the planned level of rehabilitation is also known in addition to the future output (mainly the case for modern mining activities), both can be taken into account in determining the net transformation due to the activity. This net transformation may again be either positive or negative. Thus, following the principle of distinguishing net transformation from occupation, we use the term 'net transformations' to refer to changes in land use perceived as 'not reversed' or, more loosely, 'not compensated'. This is not the same as 'irreversible' or 'impossible to compensate'.

Points of discussion
Just as metals extraction may be 'compensated' by recovery from waste, it is to be queried whether theoretical recovery processes should be taken into account, such as returning land to a near-to-natural state (as is done in the ETH database developed by Frischknecht *et al.*, 1993/1995/1996[1]). After all, if one does so in the inventory, why not also for other types of interventions (resource extractions, emissions)? If this course is not adopted, it is better to treat 'irreversible' transformations merely as factually unreversed transformations, also to avoid complex discussions as to when a transformation is truly irreversible. Estimating recovery times to arrive at fully reversed transformations, as in the ETH database (Frischknecht *et al.*, 1993/1995/1996) is therefore not recommended. It is better to estimate and account for net transformations as interventions separate from land occupation.

Impacts of occupation
One direct impact of land occupation for a particular use is to limit or even curtail the scope for other people to use that land for the period of use. Land thus becomes scarcer, leading to greater human competition for land resources. Impacts of land occupation will be discussed as a separate subcategory in Section 4.3.3.1.

Points of discussion
There is, hence, an analogy with abiotic resources (see Section 4.3.1). This analogy can be made more vivid by recalling the distinction between flows and deposits. Occupation may be seen as affecting land as a flow; transformation as affecting land as a deposit. For most abiotic resources, in addition to competition there will also be depletion. However, in the case of flows such as land (rather than deposits and funds) competition does not concern the future availability of a natural resource (which is depletion), but to the present scarcity of the stock as experienced by man[2] (see also Section 4.3.1).
Another direct impact of land occupation may be the imposition of certain characteristics on the area of land in question, at least for the duration of occupation. Depending on the reference situation to which they are compared, implicitly or explicitly, these characteristics may be perceived as positive or negative. Set off against 'average contemporary European land quality', organic agriculture is to be deemed positive, for example. From a perspective of more afforestation being better for the 'naturalness' of the European land mass or permitting greater biodiversity, though, it may be judged rather less positively. But if 'naturalness' is taken as a reference point, virtually all the land used for human activity must be deemed of 'inferior' quality. What is to be understood by the term 'occupation impact' is thus highly dependent on the defined reference situation. This issue is discussed in more detail below, in Section 4.3.3.2.
Certain forms of land occupation may, furthermore, have an indirect impact by perpetuating the stress on

[1] At present the ETH database is virtually the only LCA data source for land use data. It is based on estimates of the time involved for all transformations and occupations, and includes both occupation and (often theoretical) recovery times. In 1994, however, no distinction was made between the concept of net transformation and that of occupation. In practice, only the occupation related to a certain type of land use or occurring during a transformation process is given (in m^2.yr), and a full and instantaneous recovery to the original state is implied. Reliable net transformation data for LCA purposes is thus still lacking.
[2] As this scarcity is relevant solely to man, it might even be said that land competition impacts only on human welfare, not on the natural environment.

the natural environment associated with previous habitat destruction and fragmentation in the area or region around the land in question. In stressed areas (with high population densities) this may lead to continued loss of biodiversity (see Müller-Wenk, 1998, p.18). In this sense, occupation of land in one part of an ecosystem may render another part more vulnerable.
The indirect impacts of occupation on life support functions are less evident. Ensuing changes in the 'albedo effect' (the capacity of the earth's surface to reflect solar radiation) or in evaporation rates may significantly affect climate regulation, but as yet no assessment methods have been developed for such impacts.

Impacts of transformation
The primary impact of net transformation is a reduction of land area with a high nature quality (for instance in terms of biodiversity and life support functions[1]; see the discussion under these subcategories in Section 4.3.3.2 for a closer examination of these issues), with replacement by land of lower quality (or the reverse). Here, too, this quality depends on the desired reference quality, although one might also assess differences before and after the transformation, ignoring the reference situation. This replacement implies direct habitat destruction and attendant reduction of life support functions and biodiversity in the surrounding area, too. In terms of areas of protection, then, we are concerned explicitly with the natural environment. Large biodiversity losses in Europe are said to be caused by transformations (see e.g. RIVM, 1992).

Summary
Concluding, land occupation (in m^2.y) leads to an increase in land competition and to institution of a certain (generally low) quality for a particular period of time due to an economic process/activity and depending on the reference situation of the interpreter and selected indirect impacts. Land transformation (in m^2) changes the quality of the land itself as well as that of the surrounding area or region. Data relating to land transformation are of very poor quality and due attention should be paid to whether and how reversibility is accounted for.
The difference between land transformation and occupation shows certain parallells with that between depletion and competetive use, as discussed under abiotic depletion (Section 4.3.1).
The impact of competition for land is an impact on natural resources and, perhaps, man-made environment. The impact of decreasing quality of land can have meaning for the natural environment (ecosystems) and man-made environment. Biodiversity and life support functions are well known indicators for these endpoints. Therefore three subcategories are discerned:
- increase of land competition;
- loss of biodiversity;
- loss of life support functions

Impact assessment approaches for these three subcategories are discussed in the following two subsections: Section 4.3.3.1 for land competition and Section 4.3.3.2 for loss of biodiversity and loss of life support function.

General discussion of the SETAC WIA–2 framework
In the above description some main discussion points on the SETAC framework for land use impacts were spotted. In these discussion points some suggestions were made, which are summarised below. In the example given below much attention is given to a stricter separation of the intervention aspects and impact aspects of land use.

Description of a unit process in relation to land use
Imagine an area of land, described as being in a state A (e.g. 'forest'). At time t1 a human activity (e.g. crop cultivation) is initiated on the land, leading to a new state B. At time t2 the activity ends and a new state C emerges (e.g. grassland). We assume, provisionally, that the human activity starts and ends abruptly, and that no preparatory or clearing (up) activities are required before the human activity starts or after it ends. See Figure 4.3.3.2.

[1] Besides increase of land competition Lindeijer (2000) lists the following subcategories: degradation of biodiversity, degradation of life support functions and degradation of cultural values. The last of these is not discussed in any detail for lack of methods.

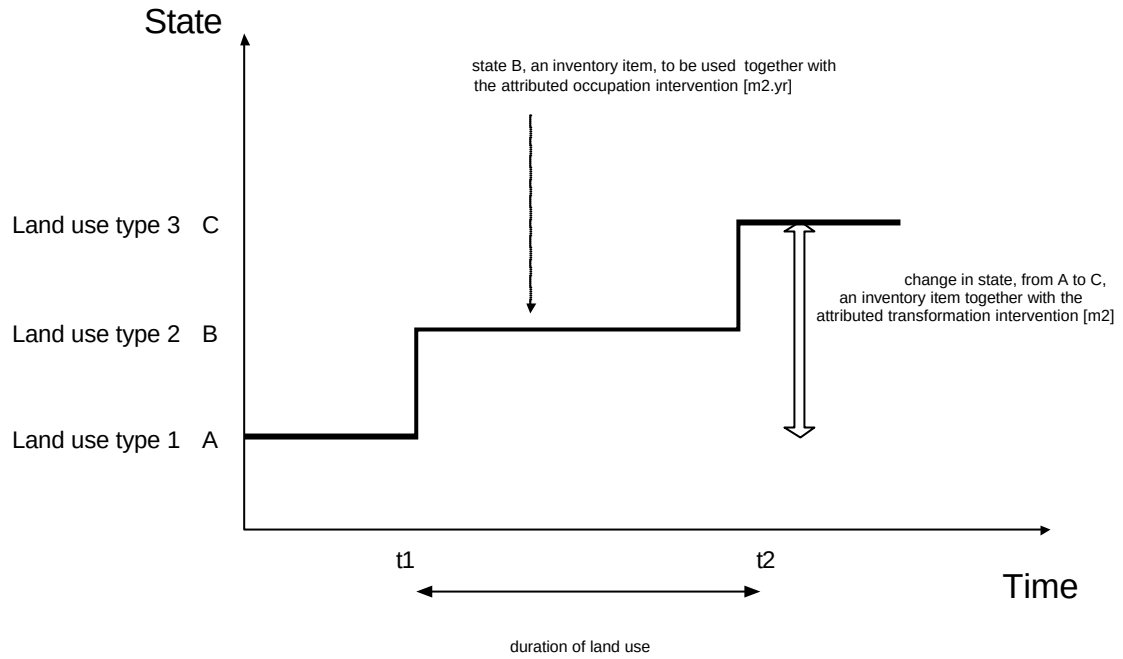


Figure 4.3.3.2: Modified version of Figure 4.3.3.1, concentrating on inventory-related issues only.

How might one describe human activity as a unit process in the Inventory analysis? Let us consider some concrete hypothetical data: a plot of land with an area of 100 square metres producing 1000 kg of crops a year, on which 1 kg/y of pesticides is used. Dividing out the year dimension, we could then attribute the following data items to this unit process:

- input of 1 kg pesticides;
- output of 1000 kg crops;
- occupation of 100 m^2·yr land in state B;
- transformation of 100 m^2 land from state A to state C.

Note that we consistently speak of state A rather than quality A, because the quality aspect is to be assessed in Impact assessment only. Lindeijer (2000) provides a similar picture, except that it is geared towards Impact assessment. Discussion in terms of state rather than quality implies that the cardinally defined quality scale in Lindeijer *et al.* has been replaced here by a nominally defined state scale, in which the order of and the difference between states makes no sense, and in which an integral is also undefined. Another difference is that the relaxation period has been added, while it has been postponed here. One final difference is that the finite duration of the human activity is essential in the discussion by Lindeijer *et al.*, while, as is shown above, an accounting of the yearly production and the land use aspects leads to sensible expressions for both the occupation of land (of state B) and the transformation of land (from state A to state C) regardless of whether the activity is of finite or infinite duration.

Attribution of land use to a functional unit

In LCA the interventions associated with the unit processes need to be attributed to a functional unit. Suppose that the functional unit here is related to 1 kg crop. The environmental interventions per kg crop are then:

- occupation of 0.1 m^2·yr land in state B;
- transformation of 0.1 m^2 land from state A to state C.

This procedure can be repeated for all unit processes in the product life cycle and will lead to many types of land use, of the occupation type (state A, state B, ..., state Z, in m^2·yr) and of the transformation type (state A to B, state B to A, ..., state Z to Y, in m^2). Interventions of identical types can then be added over the life cycle, e.g. all occupations of land of state A can be added and all transformations of land from state A to state B can be added.

Definition of impact categories

In the literature three different impact categories recur, relating to competition for land as a resource and to impacts on life support functions and biodiversity. Here, these will be referred to by the following names:

- competition;
- loss of life support functions;
- loss of biodiversity.

'Competition' is concerned with land as a scarce resource. In contrast to most types of resource, this is a resource of the flow type: every year a certain amount of $m^2 \cdot yr$ of land is available, and every activity that occupies land means there is less left for other activities. Therefore, competition is related to occupation. The other impact categories are related to transformation, as it is changes in land quality that may have implications for life support functions and for biodiversity.

The link between interventions and impact categories for land use

Note that in the SETAC framework this exclusive link between the type of intervention and the type of impact category is not made. In the suggestion presented above the intervention type 'occupation' is linked to the impact category 'competition' and the intervention type 'transformation' is linked to the impact category 'depletion in terms of loss of biodiversity and/or loss of life support functions'. In the SETAC framework and in the operational methods, which will be described in Section 4.3.3.2, both the 'transformation' and 'occupation' type of intervention are linked to 'depletion' (loss of biodiversity and loss of life support functions) (see Table 4.3.3.1). In the SETAC framework the 'occupation type of intervention' can be characterised as 'loss of biodiversity' because the framework provides a model in which the quality (e.g. in terms of biodiversity) during occupation is related to a recovery potential, i.e. the quality (biodiversity) at the final steady state if the occupation were to end immediately and the land fully recover (with or without human aid), i.e. a change from $quality_{occ}$ to $quality_{fin}$. In the suggestion made above the state during occupation is not compared to a reference state in terms of biodiversity or life support functions but is compared to the total area available in the model in order to derive characterisation factors for competition. Table 4.3.3.1 provides an overview of the methods that have been proposed for linking interventions to impact categories in the context of land use. The details of these methods are discussed in Sections 4.3.3.1 and 4.3.3.2.

Table 4.3.3.1: Overview of the relationship between land use related intervention types and impact categories, with the panoply of available methods. Observe that, given the proposed exclusive link between occupation and competition on the one hand and between transformation and depletion on the other, the cell at the juncture of occupation and depletion appears to be spurious.

		Intervention type	
		occupation	transformation
Impact category	Depletion (loss of biodiversity, loss of life support functions)	SETAC WIA–2 Lindeijer, 1998 Köllner, 2000	Proposal in this guide SETAC WIA–2 Lindeijer, 1998 Köllner, 2000
	competition	Proposal in this guide (SETAC WIA–2)	–

Conversion of interventions into impact category results

For almost all impact categories, the concept of characterisation factors makes sense. In an abstract sense, the structure of the formulae for calculating impact category results is as follows:

$$\text{result}_{\text{impact category}} = \sum_{\text{unit process}} \text{characterisation factor}_{\text{impact category,type}} \times \text{intervention}_{\text{unit process,type}} \quad (4.3.3.1)$$

Let us then apply this formula to the impact categories:

- competition;
- loss of life support functions; and
- loss of biodiversity.

There are two categories of intervention, with several types within each category:

- occupation, of state A, state B, etc.;
- transformation, from state A to state B, from state B to state A, from state A to state C, etc.

Since occupation contributes only to competition and transformation only to decrease of life support functions and decrease of biodiversity, characterisation factors must be defined for linking several types of occupation to competition and for linking several types of transformation to decrease of life support functions and decrease of biodiversity. In the following sections, some specific proposals for deriving these factors are discussed.

Consistency with other impact categories

The ISO standards stress the need for consistency among the various elements of LCA. One of the issues of concern is therefore the consistency between the classification methods adopted for the various impact categories. Including fate modeling in toxicity models but not in models of acidification is inconsistent, for example, and might bias results. This does not imply that fate should be omitted from the toxicity model but that it should, in principle at least, also be developed for acidification and that due care should be taken when employing the models as they currently stand. The category ‘impacts of land use’ differs in many fundamental respects from other impact categories. For instance, while the pathway of chemical emissions can be described in fate models, there is no such thing as a ‘pathway’ of land occupation or transformation. In certain respects, there are analogies, however. In change-oriented LCA, the starting point is a permanent marginal change of demand, with marginal effects on production and emission characteristics. In modeling the resultant impacts steady-state assumptions are then made. In discussing land use, one might now proceed from a permanent marginal change in land use required for extra production. Then, there is no t2 at which the land use stops, and there is no final state. This means that the concept of transformation is perhaps less suitable in the context of permanent marginal changes. It is clear that a consistent point-wise comparison of the models for land use and other impact categories might lead to new insights and recommendations for both groups of models. This requires a major research project, however.

4.3.3.1 Land competition

TOPIC

This subcategory of land use impacts is concerned with the loss of land as a resource, in the sense of being temporarily unavailable. The areas of protection are natural resources and the man-made environment. 'Loss of biodiversity' and 'loss of life support functions', the other two subcategories, are treated in the next section (Section 4.3.3.2).

DEVELOPMENTS IN THE LAST DECADE

It was not yet recognised by Heijungs *et al.* (1992) that all changes land use are in fact transformations from one quality to another followed by an occupation. Their formula (see textbox) can now be seen as consisting of two parts, one describing transformation (in m^2 transformed to built or degraded systems), the other occupation (in m^2.s of land occupied as built or degraded systems). Only the latter would now be regarded as leading to an increase of land competition, as well as possibly to impacts on biodiversity in already stressed areas. Transformation has an impact on both biodiversity and life support functions.

Since 1992 there has been a lot of work on the topic of land use. Most has been concerned with developing transformation and occupation indicators for various types of land use (in terms of 'naturalness'), including indicators for loss of biodiversity and life support functions (see, for example, Knoepfel, 1995; Blonk & Lindeijer, 1995; Heijungs *et al.*, 1997 and Lindeijer *et al.*, 1998). However, since the latter are treated here as separate subcategories, they will not be discussed here.

Heijungs *et al.* (1992)

In Heijungs *et al.* (1992) no distinction was made between occupation and transformation impacts, nor was the concept of competition then recognised. Land use impacts were therefore expressed in terms of m^2.s of land transformed from one land quality category to another. They recommended a provisional solution for characterising land use. Five types of ecosystems were defined: I) natural systems, II) modified systems, III) cultivated systems, IV) built systems and V) degraded systems. This was based on the *Hemerobie-stufen* concept proposed by IUCN in 1991. The land use qualities were defined according to an approximate notion of 'naturalness'. The five types of ecosystems were then aggregated into two main categories: "natural" (type I, II and III) and "non-natural" (type IV and V). This fairly arbitrary solution was considered no more than temporary. The provisional assessment was then expressed as:

$$Landuse = R_{I\to IV} + R_{II\to IV} + R_{III\to IV} + R_{I\to V} + R_{II\to V} + R_{III\to V} \qquad (4.3.3.1.1)$$

Thus all changes from I, II and III to IV and V were weighted with a factor 1, all other changes with 0. The total indicator result was expressed in m^2·s. *R* follows from the inventory and represents the quantity (in m^2·s) in each category.

With regard to the topic of land competition, developments have been sparse. Simple aggregation of land occupation, in m^2.s or m^2.y, is obviously the least sophisticated approach. When the focus is solely on competition for land as a 'resource', it would be preferable to use the same method as for abiotic and biotic depletion. This would imply use of the deaccumulation and reserves formulae or quality reduction relationships proposed for depletion of mineral resources. This approach would also enable a distinction to be made between different land qualities from the perspective of competition (thus quality is here meant 'for human use', in contrast to ecosystem quality); we would have to assess how large the resource of each land quality type is, and how fast this is decreasing or increasing[1].

There is one problem with this approach, however: there is as yet no agreement on how the 'total land resource' is to be defined. Among the options under discussion are:

- the total area of land potentially available to man, in m^2;
- the total area of land currently used by man, in m^2;
- the total area of land potentially available to man, in m^2·yr (first option multiplied by the time the land will be available to man).

Of these proposals the last is most in line with the definition for abiotic resources but it is hard to operationalise, given the likely difficulty of reaching agreement on the period deemed relevant (how long will humans populate the earth?).

[1] A priority indicator for this quality may be land fertility.

PROSPECTS

No specific developments are foreseen in this area.

CONCLUSIONS

Given the lack of agreement on a more sophisticated indicator for land competition, we recommend simple aggregation of the area of land used (in $m^2 \cdot yr$).

$$land\ competition = \sum_s U_s \qquad (4.3.3.1.2)$$

U_s [1] is the land use of state (or quality) s attributable to the functional unit, expressed in $m^2 \cdot yr$. The total indicator result is expressed in $m^2 \cdot yr$.

This method is evaluated against the (ISO-based) criteria in Table 4.3.3.1.1.

Table 4.3.3.1.1: Evaluation of the baseline characterisation method for land competition, characterisation factor = 1, with respect to the (ISO-based) criteria of Table 4.3.1.

	criteria	evaluation
1.	scientifically and technically valid	not relevant
2.	environmentally relevant	category indicator is close to intervention
3.	internationally accepted	no
4.	value-choices and assumptions	no
5.	focal point in environmental mechanism	close to intervention
6.	linearity	yes
7.	time span	not relevant
8.	fate, exposure/intake and effects	not relevant
9.	less is better	yes
10.	time- and location-independent	yes
11.	operational	yes
12.	uncertainty margins	none

No methods are included in the Guide as options for sensitivity analyses.

No additional methods are recommended for extended LCAs.

characterisation method	characterisation model	reference
baseline	unweighted aggregation	Heijungs *et al.*, 1992
alternative	–	–
additional	–	–
variant	–	–

RESEARCH RECOMMENDATIONS

Long-term research:

- It might be useful to develop a method to account for competitive use of flows (and funds) and investigate the potential for aggregating flows, deposits and funds, including the impacts of land occupation on land competition. Research is recommend to examine how the difference between competitive use and depletion might be accounted for in LCA and how land competition might be included in the category 'abiotic depletion' (see also Section 4.3.1).
- It might be interesting to try to incorporate the differences in the quality of the land that is occupied. However, this would require a definition of the 'reserves' of each land use type, at least if formulae similar to those discussed under abiotic depletion are used. Besides the problem of defining the reserves in m^2 or $m^2 \cdot yr$, other problems are to be resolved, for instance, relating to the effects of national land use policies in determining the degree of competition.

[1] Note that the composite notation $A \times t$ is sometimes used in the literature instead of U. This suggests, however, that the area A and time t attributable to a functional unit are known separately, while in fact only the composite quantity is known.

4.3.3.2 Loss of biodiversity and loss of life support functions

TOPIC

These two subcategories are discussed here together in one section because this is generally the case in the literature and it is difficult to separate the strands of the associated discussions. The topic encompasses the entire range of impacts on biodiversity and life support functions of physical interventions due to a particular form of land use, involving destruction or alteration of land for economic purposes, for example, or the harvesting of biotic resources, probably in areas already under biodiversity stress. These two subcategories have the natural environment and, indirectly, natural resources as areas of protection.

DEVELOPMENTS IN THE LAST DECADE

Methods for dealing with loss of biodiversity and life support functions as a result of land use or use of

> Heijungs *et al.* (1992)
> These impact categories were not included in Heijungs *et al.* (1992).

biotic resources in LCA have been put forward in many recent publications, including Knoepfel (1995), Lindfors *et al.* (1995c), Blonk & Lindeijer (1995), Sas *et al.* (1996), Heijungs *et al.* (1997), Lindeijer *et al.* (1998) and Köllner (2000). The methods proposed by the various authors fall broadly into three groups, as described below (mainly from Heijungs *et al.*, 1997 and Lindeijer *et al.*, 1998).

1. *Ecosystem classes*
   Classes of ecosystem are distinguished and transformations from one class to another and the occupation of each class are collected in the inventory. The classes are then weighted. Knoepfel *et al.* (1995) and Lindeijer *et al.* (1998) present overviews of such classifications, including weighting methods (based on panel preference, biodiversity or other properties of the land). Beetstra (1998) has added a monetary weighting method to the classes proposed by Knoepfel *et al.* (1995).
2. *Biodiversity and life support function indicators*
   Examples are the last two formulae of Sas *et al.* (1996), described in Section 4.3.2 on biotic resources. Lindeijer *et al.* (1998) propose a similar approach in which the (change in) free Net Primary Production (fNPP; i.e. the Net Primary Production minus the Net Primary Production for human use, where Net Primary Production is defined as the net increase in the dry weight weight of plants) or species density is compared with a reference situation. Köllner (2000) has elaborated the basic biodiversity indicator to include impacts on the region surrounding the assessed area and a marginal assessment of indicator results.
3. *Functional aspects*
   Baitz (1998) has developed a method in which several characterisation factors are calculated, for the quality of an area before, during and after an intervention. These are based on the capacity of the land to fulfil ecosystem and human life support functions and are expressed in physical terms. The method has been elaborated for a single caseat a specific location. In this method a set of eight characterisation factors are proposed for land use. These can be calculated for concrete local situations and will in the future be able to be estimated for generic processes using GIS systems and other readily available data. Such generic information is not presently available, however, nor is a set of local data suitable for extrapolating to the generic level. Schweinle (1998) has developed a similar approach for the forestry sector, but omits the area parameter in his Impact assessment formulae.

It is not clear what the relationship is between these three groups and the aforementioned three types of indicators for biodiversity.

It can be expected that the methods designed to describe loss of biodiversity and life support functions (group 2) are presently the best available practical methods for these impact categories. Some of the indicators using classes (group 1) are also based on biodiversity or life support function parameters (e.g. species density and NPP). However, a disadvantage of any method based on classes is that it can only describe rough categories of land use. Such methods do not permit detailed description of changes in biodiversity or life support functions, nor do they allow for regionalisation. The methods based on functional aspects (group 3) are not aimed specifically at biodiversity or ecosystem life support functions, but at a large number of mid-point indicators. Moreover, there is currently insufficient data available for this method to be operationalised. Concluding, we shall focus on the methods considering

loss of biodiversity and life support functions (group 2), these being the most appropriate for present characterisation of loss of biodiversity and life support functions.

Of quite a different nature is the approach to loss of biodiversity adopted by Anonymous (1997b; 2000a; 2000b). Two pressure indicators are defined: one for the environmental pressure on the area of interest due to the present activity and one for general pressure on the region. It is a nominal approach having a large overlap with other impact categories such as ecotoxicity and eutrophication. Moreover, it assumes a fairly detailed knowledge of local aspects and requires data on the previous history of land conversion. Last, the concept of characterisation factors does not appear to be applicable. All these features lead us to conclude that it may be an interesting approach for certification but is not presently suitable for application in LCA.

Finally, the connection with other types of impact such as biotic depletion, desiccation and ecotoxicity should be mentioned. These impact categories are discussed in separate sections (see Sections 4.3.2, 4.3.4 and 4.3.8), but it is clear that biodiversity and life support functions are closely related to these other impact categories.

To the best of our knowledge, three methods are currently available for biodiversity and life support functions. All three are based on the category indicators plant species density and Net Primary Production (NPP) of biomass for specific regions or ecosystems. These methods have been developed by Sas *et al.*(1996), Lindeijer *et al.* (1998) and Köllner (2000). For the Eco-Indicator 99 Goedkoop & Spriensma (1999) use a slightly adapted verison of Köllner (2000). Lindeijer *et al.* (1998) have reviewed all land use methods developed to date and included Köllner (2000) and others in a later review (Lindeijer, 2000).

Some general aspects of biodiversity and life support functions and their indicators
Before examining the methods available for life cycle impact assessment of loss of biodiversity and loss of life support functions, it is appropriate to look at these two types of loss in further detail in a more general sense, i.e. not in relation to LCA

Indicators for biodiversity
In the literature three aspects of biodiversity are generally distinguished:
- type of species (the intrinsic value of a unique genetic code);
- type of ecosystem (a unique combination of types of species and their interrelationships);
- species density (the number of different species per unit area, independent of the type of species or ecosystem).

On this basis, the loss of biodiversity in a given area might, in principle, be expressed in terms of three, related category indicators: loss of species, loss of types of ecosystem and reduction of species density. These indicators require information on overall species and ecosystem diversity, present occurrence (at least biomass) and current rate of disappearance. There is a paucity of knowledge on all these elements, however, and for this reason 'loss of biodiversity' is frequently expressed solely in terms of reduction of species density. However, this is a highly abstract indicator that says nothing about the types of species and types of ecosystems that are vanishing.

Indicators for life support functions
For life support functions, too, a variety of indicators are conceivable. That most commonly used is Net Primary Production (NPP), i.e. the net increase in the dry weight of plant matter. The assumption then made is that if current biomass production on an area A is high, then soil quality will be stable, freshwater cycling intact and rainfall adequately absorbed by the soil. However, high biomass production under the influence of man (agriculture and silviculture) may be associated with high rates of erosion and low quality freshwater run-off. Due consideration should therefore be given to the fraction of biomass remaining to support non-human life forms. This may also give an indication of the long-term capacity for life support. An alternative parameter is *free Net Primary Production fNPP* (also referred to as *biomass appropriateness*), i.e. total NPP minus the portion of NPP removed from area A as an agricultural or silvicultural product.
Table 4.3.3.2.1 reviews the characteristics of the three methods, which are briefly described below.

Table 4.3.3.2.1: Characteristics of three operational methods to assess impacts on biodiversity and life

support functions.

method	type of intervention		category indicator		correction factor	operational for regions?
	Transf. [m^2]	Occ. [m^2.yr]	biodiversity	life support functions		
Lindeijer *et al*.	+	+	plant species density	fNPP		world climate zones
Sas *et al*.		+	species density	NPP	t_{recov}	ecosystems; sea and forest
Köllner	+	+	plant species density		regional impacts	Swiss lowland

NPP Net Primary Production
*f*NPP free Net Primary Production = NPP minus NPP for human consumption
t_{recov} recovery time to the original biodiversity or biomass, a weighting factor expressing the vulnerability of the ecosystem.

Method of Lindeijer *et al*.
The method of Lindeijer *et al.* (1998) operationalises one indicator for biodiversity and one for life support, and makes a distinction between occupation and transformation. The following formulae are used:[1]

For net ecosystem transformation, the change in the quality of an area of land following recovery from a land use intervention is compared with that prior to that intervention:

$$loss\ of\ biodiversity = A \times \left(\frac{a_{ini} - a_{fin}}{a_{ref}} \right) \quad (4.3.3.2.1)$$

$$loss\ of\ life\ support\ functions = A \times \left(fNPP_{ini} - fNPP_{fin} \right) \quad (4.3.3.2.2)$$

For ecosystem occupation the quality during occupation (including recovery) is compared with the quality in the reference, i.e. unoccupied state:

$$loss\ of\ biodiversty = A \times t \times \left(\frac{a_{ref} - a_{act}}{a_{ref}} \right) \quad (4.3.3.2.3)$$

$$loss\ of\ life\ support\ functions = A \times t \times \left(fNPP_{ref} - fNPP_{act} \right) \quad (4.3.3.2.4)$$

In which:
A is the area of land used,
t is the occupation time[2],
fNPP is the free Net Primary Production[3],
a is the number of plant[4] species per m^2, calibrated to a reference measurement area[5].

[1] Note that most of the formulae that follow, from Lindeijer *et al*., Sas *et al*. and Köllner, have no summation over different types of land, ecosystems, species, etc. although such summation will in most cases be intended. Another feature is that Lindeijer *et al*. and Sas *et al*. do not explicitly introduce characterisation factors, while Köllner does so in the form of the SPEP.
[2] Note that in the impact assessment of an LCA *A* is the attributed area of the land use activity to the functional unit and *t* is the attributed time period. These two quantities are often not known separately, but only in composite form.
[3] There is however still some uncertainty as to how fNPP is determined (especially as this measure is relatively high for tropical rainforests, which is unexpected for the Net Primary Production in an ecosystem in equilibrium).
[4] The general indicator on the basis of species density is thus approximated by an indicator on the basis of plant species density.
[5] This calibration is necessary because the area-species relationship is not linear. When biodiversity mapping data are used to express the reference, initial, actual or final situation all these data should be calibrated to the same standard (reference cell) on this curve. For this procedure the following formula is used: $S_{map} - S_{ref\ cell} = \alpha * \log(A_{map}/A_{ref\ cell})$. For the small changes considered in LCA linearity is assumed. In other words: the land change assessment method is only valid for non-critical changes (such as the last area available for rhino's). Also, only local biodiversity changes are assessed; the area around the land use change is in first instance assumed to allow for reversibility of the local impact in principle.

Assume two interventions with the same magnitude of area (e.g. 100 m^2).

Starting from a natural situation, changing from 100 to 20 species per m^2 in a tropical rainforest is considered equal to changing from 10 to 2 species per m^2 in boreal rain forests. A change of 8 species per m^2 in tropical forest will cause less impact than a change of 8 species per m^2 in boreal forest. A same relative change as above due to industry in former intensive agricultural land (e.g. from 5 to 1 species per m^2 yields a lower impact than coming from a natural situation (i.e. 10 species per m^2). This is because for an impact due to transformation the initial situation is expressed relative to the most natural situation. Meaning that a relative change in a non natural situation, e.g. intense agriculture, is considered less problematic than a same relative change in a natural ecosystem.

ini	stands for initial, the situation before the intervention,
fin	for final, the situation after the recovery period, when the intervention has stopped
act	for actual, the situations during the intervention and
ref	for reference, the present most 'natural' situation in the region where the land use takes place (see also Figure 4.3.3.2.1).

In Figure 4.3.3.2.1 the model for the calculation of the transformation and occupation impact according to the method on Lindeijer *et al*. (1998) is illustrated. In this figure also the proposed method by SETAC WIA–2 (Lindeijer, 2000) is given.

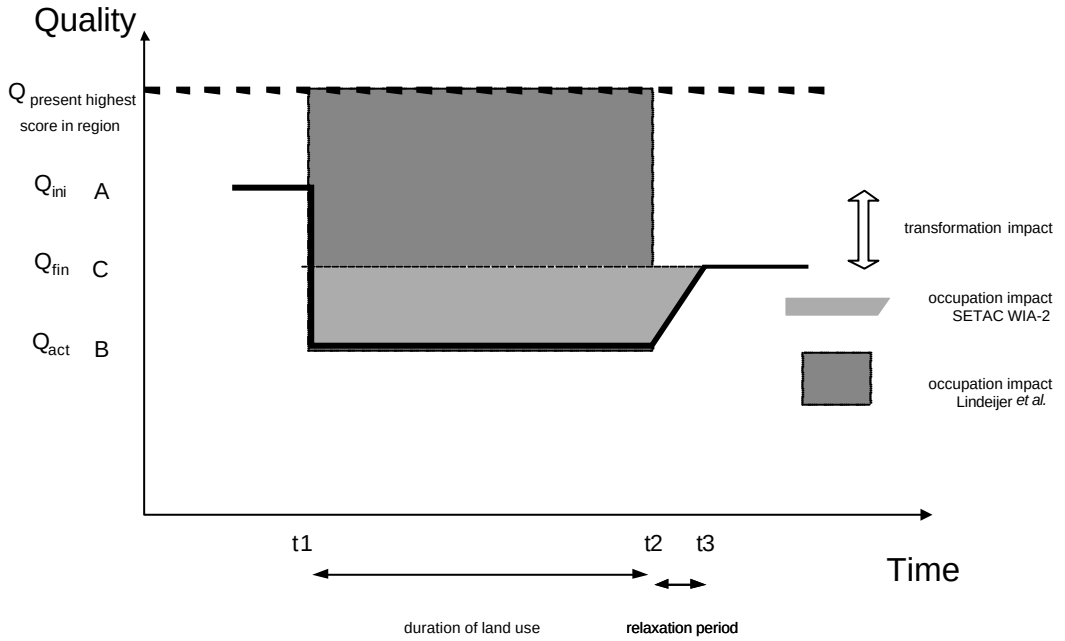


Figure 4.3.3.2.1: The transformation and occupation impact for a given area (y m^2) of land use according to the method of Lindeijer *et al.* (1998). The quality Q is expressed for the subcategory biodiversity by the plant species density (*a)*, and for the subcategory life support functions by the free Net Primary Production (fNPP).

The method for ecosystem occupation accounts for occupation time, the method for ecosystem transformation only for net changes. The method for occupation relates the actual situation (a_{act}) to the reference situation(a_{ref})[1]. The method for transformation relates the changed situation (a_{fin}) to the situation before change (a_{ini}). For biodiversity, expressed in terms of plant species density, the absolute terms are expressed as relative to the reference state to correct for spatial differentiation of plant species density between regions, due to differences in ecosystems and climate zones etc., all over the world. So changes in plant species density are expressed relative to a reference to ensure the intrinsic value of

[1] Note that this definition is not in line with the proposal of the SETAC WIA–2, which has proposed to use (Q_{fin} - Q_{act}), instead of ($Q_{ref} - Q_{act}$) to derive the occupation impact.

each ecosystem type The reference is defined as the highest species density currently found in the region (generally). The region is on the level of physiotopes, large regions related to climate zones on a world level.

In the approach of Lindeijer *et a*l. aggregation of occupation and transformation is discouraged, as this would require the assumption that all land transformations are reversed within a certain (to be estimated) recovery time. The method for occupation thus focuses on the maintenance of biodiversity and biomass

Remarks on the reference situation

Lindeijer *et al.* (in prep.) provide no guidelines on a choice of reference situation. Should this be at the level of physiotopes (tropical, boreal, etc.), continents (Europe, Asia, etc.), countries (Netherlands, Switzerland etc.) or ecosystems (heathland, meadow, forest)? Should it be the maximum biodiversity of the region, the near-natural situation in the absence of human activity or the maximum biodiversity in recent history (for Europe around 1900)?

As yet, the choice of the reference situation thus seems rather arbitrary. For biodiversity, measured in terms of species density, Lindeijer (1998) has chosen to take the "maximum biodiversity of the region", with the region defined in terms of its physiotope (tropical, desert, boreal, etc.). This is contrast to Köllner (2000), who opts for the maximum biodiversity in recent history at the national level (for Switzerland around 1900).

Note that the indicator "species density" says nothing about loss of unique species or ecosystems. For example, consider an activity on heathland (an ecosystem with few species, say 5 species/m^2, a level artificially maintained below the level to be expected in a near-natural state in this geographical region, say 15 species/m^2). After discontinuation of the activity the area will return to a new steady state. In terms of species density this final state is likely to be closer to the near-natural state then the initial state of the heathland (say 10 species/m^2). The transformation impact for the impact category "loss of biodiversity" based on species density is therefore –1/3 (i.e. 5 minus 10, divided by 15), which is a negative value, signifying an increase in biodiversity (read species density). Although the activity is thus beneficial for the number of plant species, then, an ecosystem is lost (viz. heathland) and the intrinsic value of that ecosystem is therefore not respected. For the same reason, conversion of an area of desert (low species density) into a meadow (higher species density) would be considered beneficial for biodiversity (represented by plant density), although the overall impact is in fact destruction of a desert ecosystem.

characteristics due to land occupation relative to a near-natural reference. Indirect impacts on biodiversity via stress on the surrounding environment is not assessed.

Lindeijer *et al.* (1998) have calculated characterisation factors for seven combinations of land use and region. However, the authors stress that the data on fNPP and α after the intervention (=act or fin) are rough estimates based on rather scanty data. For other combinations of land use and region, characterisation factors must be developed case by case. The α and fNNP in the reference situations for different regions worldwide required for calculation of these factors are already available in the form of two maps and two tables. These maps are based on empirical data and expert judgement. Data on α and fNNP resulting from land use other than the seven types mentioned (Lindeijer *et al.* state that these data on α and fNNP after intervention are generally applicable) should be collected on a case by case basis. This may be very time consuming. On the other hand, Lindeijer *et al.* state that these seven types of land use are the most important and that most LCAs can be satisfactorily performed using the data (see also under data availability).

Method of Sas *et al*.

The method of Sas *et al.* was originally developed for biotic depletion and is described in Section 4.3.2. The formulae used there for biodiversity and life support impacts on ecosystems were (rewritten for this section):

$$\text{loss of biodiversity} = A \times t_{recov} \times S \qquad (4.3.3.2.5)$$

to indicate the risk on the extinction of species by destroying an ecosystem and

$$\text{loss of life support functions} = A \times t_{recov} \times NPP \qquad (4.3.3.2.6)$$

to indicate the loss of life support functions,

with:

t_{recov}	recovery time to original biodiversity or biomass,
S	initial species diversity for a fixed area and NPP (not subtracting the amount of biomass used by humans, as in fNPP).

Remarks on Net Primary Production

Not excluding human biomass consumption makes a big difference, for instance for commercial forestry. The majority of biomass in silviculture is wood which will not contribute to the carbon cycling via natural degradation; other contributions to substance cycles are either taken into account in good LCAs (for instance for CO_2) or are disturbed by the forestry activities. Therefore, we consider NPP not as good an indicator for life support as fNPP.

This method implicitly only assesses transformation and not occupation. The recovery time is here a weighting factor, expressing the vulnerability of the ecosystem. It is different for S and for NPP. The method of Sas *et al.* (1996) is only partly operational. They give some rough data on species density, NNP and regeneration time for forest ecosystems and a marine ecosystem.

Method of Köllner

The method of Köllner (2000) assesses only biodiversity and not life support functions. The category indicator used for biodiversity is plant species density. Besides transformation and occupation the method also incorporates a correction factor for regional impacts of the land use in the region surrounding the local area where the actual intervention takes place.

The characterisation factor for the impact on biodiversity is SPEP, which stands for Species-Pool Effect Potential. Each type of land use has its own $SPEP_{local,\ occ}$, which is based on the ratio between the number of plant species found in the area occupied for a specific type of land use and the average number of plant species found in the region as a reference. The relationship between SPEP and the relative number of species for a land use type is described as a negative log-curve. The method is thus a marginal approach, determining the slope of a species-area curve for each land use type.

Characterisation factor for occupation

The characterisation factor for local occupation through an activity associated with a specific land use type is defined as follows:

$$SPEP_{local,occ} = -a \times \ln\left(\frac{S_{landuse}}{S_{region}}\right) - b \qquad (4.3.3.2.7)$$

with:

$SPEP_{local,occ.}$	Species-Pool Effect Potential on the local scale for occupying a specific land use type
$S_{land\ use}$	species number on specific land use type (e.g. industrial area, intensive meadow)
S_{region}	average species number in the region (reference).

The parameters a and b are set to a = 0.8 and b = 0.9 to get a SPEP curve which has the following characteristics:

- SPEP is 0 when about 30% of the regional species richness is present on the specific land use type. This type of land use is assumed to be a moderate condition.
- SPEP is 1, when about 10% of regional species richness is observed for the specific land use type. This type of land use is assumed to be a bad condition.
- If less than 30% of the regional species are found for the land use type the SPEP is positive, indicating damage. If more than 30% of the regional species are found for the land use type SPEP turns negative, indicating benefit.
- The SPEP curve steeply decreases for types of land use with less than 30% of the regional species and smoothes out for types of land use with more than 30% of the regional species. The log-function arises from references stating that the relationship between species richness and ecosystem functions have this form.

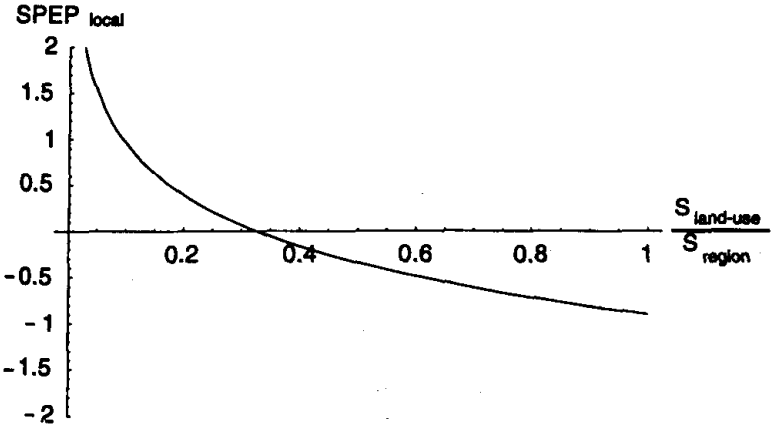


Figure 4.3.3.2.2: $SPEP_{local}$ is a function of the relative number of species for a specific land use type.

Characterisation factor for transformation
For transformation of a specific land use type I into a land use type II a $SPEP_{transformation}$ can be defined by relating $SPEP_{occupation}$ of land use type I to $SPEP_{occupation}$ of land use type II. Land use types I and II are nominal classes, e.g. intensive meadow, organic meadow.

The local effect factor for transformation is defined as:

$$SPEP_{local,transf.I\rightarrow II} = -SPEP_{local,occ.I} + SPEP_{local,occ.II}$$

$$= 2b + a \times \ln\left(\frac{S_{landuseI}}{S_{region}}\right) - a \times \ln\left(\frac{S_{landuseII}}{S_{region}}\right) \qquad (4.3.3.2.8)$$

with:

$SPEP_{local,occ.}$	Species-Pool Effect Potential on the local scale for occupying a specific land use type
$SPEP_{local,trans.\ I\ ->\ II}$	Species-Pool Effect Potential on the local scale for transforming land use type I into land use type II
$S_{land\ use}$	species number on specific land use type (e.g. type I or II)
S_{region}	average species number in the region (reference)

Weighting factor for regional impact
The local effect factor is weighted with a regional effect factor, which is the same for occupation and transformation. The reasoning behind the weighting factor proceeds from the assumption that, for example, transforming natural forests is more damaging to the region if there are only few km^2 in the region left and less damaging to the region if large areas in the region are covered with natural forests. So a relative high intensity land use in the surrounding region makes the region more vulnerable for the local interventions. As a result, greater weight should be attached to the local intervention.

The regional effect factor again considers marginal changes, given as the derivative of a species-area relationship[1] in an area with low land use intensity (LI), calibrated with the species number:

$$SPEP_{regional} = \frac{\frac{dS_{region}}{dLI}}{S_{region}} = \frac{abLI^{b-1}}{aLI^{b}} = \frac{b}{LI} \qquad (4.3.3.2.9)$$

with:

$SPEP_{regional}$	Species-pool Effect Potential on the regional scale
S_{region}	average species number in the region (reference) = a * LI^b
dS_{region}	first derivative of the equation to derive S_{region}
dLI	LI is the proportion of low-intensity land use in the region

Thus, as the proportion of high-intensity land use in the region increases, LI will decrease and as the correction factor therefore increase. Relatively high-intensity land use in the surrounding region therefore renders the regional area more vulnerable.

The occupation impact of a land use activity is now defined as:

$$E_{occ} = A \times T \times (SPEP_{total,occ.}) = A \times T \times (SPEP_{local,occ.} \times SPEP_{regional}) \qquad (4.3.3.2.10)$$

and the transformation impact of a land use activity as:

$$E_{transf} = A \times (SPEP_{total,transf.I \rightarrow II}) = A \times T \times (SPEP_{local,transf.I \rightarrow II} \times SPEP_{regional}) \qquad (4.3.3.2.11)$$

with:

$E_{occ.}$	potential impact of land occupation (hectare·year)
$E_{transf.}$	potential impact of land transformation ([hectare)
A	area occupied or transformed (hectare)
T	duration of occupation (year)
$SPEP_{local, occ.}$	Species Pool Effect Potential for occupying a specific land-use type on the local scale
$SPEP_{local, transf. I ->II}$	Species Pool Effect Potential for transformation of land-use type I into type II on the local scale
$SPEP_{regional}$	Species Pool Effect Potential for the regional scale

Data were gathered on $SPEP_{local}$ and $SPEP_{regional}$ for 16 land use types, taking the average species richness of the present Swiss Lowlands as a reference. Some land use types score positively, to be interpreted as a positive effect on (present) species richness. The main advantage of Köllner's method is that it appreciates the marginal nature of impacts on biodiversity.

[1] Instead of the log-function used for the species-area function in Lindeijer *et al.* (1998), Koellner (1999) uses the Arrhenius formula: $S = a^* A^b$. In this formula a is a parameter for species richness and b is a parameter for species accumulation rates. These are different parameters from the a and b used in the formulae for local land use impacts. These a and b can be in the range of 10 to 270 and 0.1 to 0.5 (0.9 for continuous urban land use) respectively.

Comparison of the transformation and occupation impacts of SETAC-WIA and Köllner

In Figure 4.3.3.2.3 the occupation and transformation impact according to Köllner and SETAC-WIA (Lindeijer, 2000) are compared. The quality Q represents the biodiversity indicator, viz. plant species density. In Köllner's method the SPEP of a specific land use type is 0 when about 30% of the regional species richness is present on the specific land use type. This type of land use is assumed to be a moderate condition. So implicitly for the occupation impact an actual quality due to a certain land use type (in Figure 4.3.3.2.3, land use type II) is compared to a moderate quality, defined as 30% of the average quality in the region. In this respect Köllner's method differs from that of SETAC WIA–2, which states that the impact should be based on the difference between the actual level (level B) and the level of the current recovery potential, i.e. the final steady state if the occupation were to end immediately and the land recover (with or without human aid) (level C). In other words, Köllner assumes one final steady state independent of the type of actual land use from which the area should recover and independent of the initial type of land use before the intervention took place. As shown in Figure 4.3.3.2.3 the definition of the transformation impact also differs from the proposal of Lindeijer (2000).

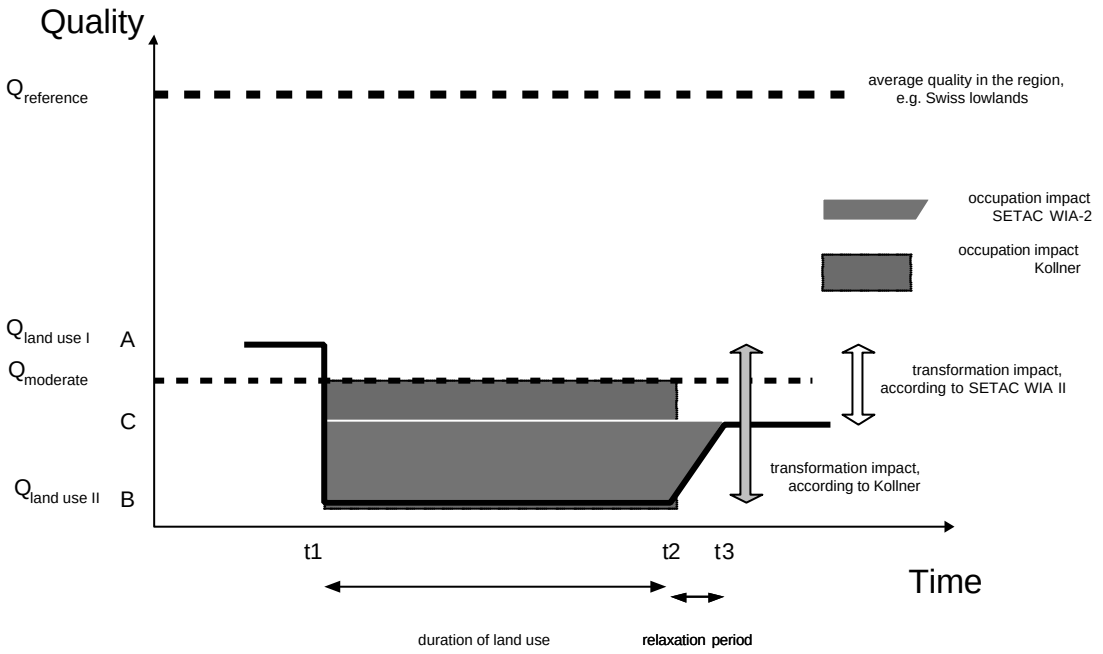


Figure 4.3.3.2.3: Transformation and occupation impacts for a given land use intervention (y m^2) according to the method of Köllner (2000) compared to the proposals of Lindeijer (2000).

Regionalisation

Potential for applying the method on a global scale is a general starting point for the selection of category indicators (see Section 1.2.2.3 and Section 1.2.3.3). However, for the assessment of the impact of land use on biodiversity and life support functions regionalisation might well be inevitable. In a global (rather than regionalised) LCA it would be necessary to employ an α and fNNP for the 'average' global ecosystem. Using the maps produced by Lindeijer *et al.* (1998) this would already introduce a major variation: α_{av} = 24 ± 15 and NNP_{av} = 8.7 ± 5,6 ; most land use transformations would fall within this variation. In fact, the data used by Lindeijer *et al.* (1998) to arrive at this average already contain an uncertainty of 50–100% due to natural variability. This suggests that it may be of no value to try to assess land use impacts in the absence of regionalisation.

In Köllners method regionalisation is not yet operational; at present the Swiss Lowlands constitute the sole reference, i.e. no reference values are yet available for other regions of the world.

Data availability

The methods of Lindeijer *et al.* (1998) and Sas (1996) are based on a regional LCA approach and need a regionalised inventory. The latter method only assesses transformations and requires very specific data.

It is necessary to know in which ecosystem the land is used. The method of Lindeijer *et al.* (1998) allows for a rough but worldwide regionalisation for occupation, operationalised with maps on reference data. For the inventory, this means that it should be known in which of the four regions of Europe European production processes, say, take place; generally the mid-European region can be taken as baseline. In general, by taking into account which type of activity (process) is responsible for the land use, the actual state for occupation can be taken into account. The reference state can only be taken into account in a general way, by assessing where on earth which percentage of the activity takes place. In Lindeijer *et al.* (1998) this was done roughly for most mining, agriculture and silviculture activities.

Köllner (2000) has published at least two articles that include characterisation factors for over ten land use types. These data have been used in the Eco-Indicator 99. However, at present the EI 99 database does not yet fully include land occupation (status March 2000). The IVAM ER database has incorporated land occupation data for all processes based on the method of Lindeijer *et al*. (1998). The land occupation data has also been translated into land use types used in the Eco-Indicator 99 method, thereby allowing the IVAM ER database also to be assessed with the method of Köllner (2000).

For transformation data on the initial and final states are also required. This has only been attempted in the ETH database (Frischknecht *et al.*, 1993/1995/1996), and then in a very generic and non-transparent fashion. In fact, no meaningful transformation data can be extracted from this database. This means that no transformation data are available, other than the data for several specific cases mentioned in Lindeijer *et al.* (1998). Transformation data will therefore have to be gathered on a case-by-case basis.

Issues to be addressed

The above formulae have been derived from the concept presented by SETAC WIA–2. Note, however, that this concept is still under development and that many issues are still the subject of discussion. Some of these issues are presented below.

1. The absolute change in plant density is expressed as relative to the reference state. The motive is to correct for spatial differentiation of plant species density between regions, due to differences in ecosystems and climate zones, etc., all over the world. However, the question is: how detailed should this spatial differentiation be: ecosystems, nations, continents, climate zones? Another issue is what quality of the region is to be taken as a reference: the situation 100 years ago, the natural climax vegetation, …?
2. In practice, derivation of the initial and final quality of the area may also be rather problematical. Only the actual quality of the area will generally be known and assumptions will therefore have to be made on the situation prior to land use. Furthermore, little information is available on the quality level to which an area will recover after a certain type of land use in a certain type of region has stopped. In practice a comparison of the actual state with the reference rather than the final situation might well be the only feasible option. This would mean that the presently operational method of Lindeijer *et al*. (1998) is practically the best feasible option.
3. Is the present indicator ‘plant species density’ a representative indicator for biodiversity? After all, by focusing solely on plant species it ignores other important elements of biodiversity, such as other (types of) species and types of ecosystems. Furthermore, it would be theoretically preferable to use, instead of only plants, a set of species picked from the whole taxonomic system with a view to better representation of species. If more indicators are chosen, how should the different indicators be weighted to one score for biodiversity?
4. Is the present indicator ‘free Net Primary Production’ a representative indicator for life support functions? Alternative indicators have been derived based on soil properties, e.g. organic matter content and physical properties (Mattson *et al*., 2000; Baitz, 1998; Milài Canals *et al*., 2000). Again, the question is: should more indicators be used to represent the impact category ‘life support functions’? Or is there one indicator that can be used to represent the whole?
5. There is an interaction between the biodiversity on a particular plot of land and that in the region surrounding the plot. For life support functions, too, this kind of interaction is to be anticipated. The interaction works both ways. Ideally, the mechanisms of interactions between the local area and the region should be part of the characterisation model. However, scientific data for a quantitative link are unlikely to be available. At the present time, only Köllner (2000) has attempted to assess the impact of a change in biodiversity in a local area on that of the surrounding region.

6. Even some of the inventory-related issues have not yet been solved. For instance, there are still controversies surrounding the definitions of occupation and transformation and the feasibility of connecting these notions to a functional unit.

Summary of review

- There is no single 'authorised' method for assessing the impacts of land use in terms of loss of biodiversity and loss of life support functions. However, a conceptual scheme for land use impacts is currently being developed and debated (Lindeijer, 2000).
- There are two operational methods for assessing impacts on biodiversity (indicator: plant species density) due to land occupation and land transformation: Lindeijer *et al.* (1998) and Köllner (2000).
- There is one operational method for assessing impacts on life support functions (indicator: free Net Primary Production) due to land occupation and land transformation: Lindeijer *et al.* (1998).
- For both methods, assessment of the impact of occupation is not yet in line with the proposals of SETAC WIA–2.
- Only the assessment of the impact of transformation according to Lindeijer *et al.* (1998) is in line with the proposals made by SETAC WIA–2.
- Both methods are presently very limited in terms of the number of land use types that can be assessed; for other land use types characterisation factors must be developed on a case-by-case basis.
- Regionalisation is required to assess the biodiversity and possibly life support impacts of land use.
- At present, only the method of Lindeijer *et al.* (1998) includes a first rough regionalisation of the reference values required for calculating the characterisation factors for biodiversity (plant species density) and life support functions (fNPP); Köllner's method is based on the Swiss lowlands and might, at best, be representative merely for central Europe.
- Regional data on actual, initial and final quality situations of land use, required for calculating the characterisation factors for biodiversity (plant species density) and life support functions (fNPP) are generally lacking and will have to be gathered on a case-by-case basis.
- The SETAC-WIA framework for land use and its impacts can only be considered as a first step in tying together the various different approaches. At present, it has insufficient conceptual clarity and has a number of inconsistencies.

PROSPECTS

Besides loss of biodiversity and loss of life support functions there are also other impacts that used to be filed under 'land use', such as landscape degeneration and desiccation. At the moment there are no known indicators for these impacts that might be used in LCA[1]. In 1999 a research project was started by RIZA and KIWA (NL) to study the possibility of incorporating an indicator for desiccation in LCA. A follow-up project to operationalise desiccation, using Lindeijer's and/or Goedkoop & Spriensma's method, is scheduled to start by mid–2000. A more extensive discussion on desiccation can be found in Section 4.3.4. Vascular plant species richness is used as a basic indicator for biodiversity impacts of land use. As mentioned in Lindeijer *et al.* (1998) and Köllner (2000) this implies that plant species are good indicators for total diversity; this may be inadequate for conservationists. If data permit, this indicator might be weighted according to rare species or rare habitats. Eventually, vertebrate richness might also be included as birds and mammals are included in global biodiversity plans. However, this would probably lead to an adaptation of the formulae and weighting of vertebrates and plants would be required. Finally, impacts of occupation on biodiversity may be excluded for non-stressed areas if it can be proven that no significant biodiversity impacts occur due to mere occupation of land (i.e. with no change in quality) in these non-stressed areas.

Species richness can be accompanied by other indicators for biodiversity changes to assess and state

[1] Early attempts to assess landscape degradation in terms of top-height (see Knoepfel, 1995) or above-ground biomass (see Lindeijer *et al.*, 1998, p. A1.27) are not acceptable for all land use types (see Lindeijer *et al.*, 1998, p. 20). For landscape degradation, the only possible indicator which seems to have some uniform value is landscape diversity. This implies that only combinations of different types of land use can be assessed. Moreover, changes in one land use type within such a local combination of land uses, or occupation by this combination can only be assessed by means of panels. Combination of panel results from different regions to an overall LCA view on the landscape degradation seems impossible, both conceptually and practically.

the total impact of land use on the natural environment. The biomass indicator for life support functions might also be accompanied by others, like soil quality. The functional approach (see under Developments in the last decade) could serve as a framework for this broadening of the set of indicators. However, clear links should be made to the endpoints for each indicator. This calls for quantification of the indicators, the endpoint and ideally the relationship between the two. This is not yet within the bounds of scientific knowledge. Again ideally, the relationship between biodiversity and life support functions should also be quantified. This is not currently feasible and, indeed, may never be. Finally, irreversibility of transformations should be considered in contrast to potential impacts due to occupation, taking into account different types of endpoint impacts.

Another line of improvement is (more detailed) regionalisation. This relates not only to the reference state, but also to initial and final states of transformations. If regional changes in land use can be attributed to specific land use types, this would allow a generic inclusion of transformation data per region. Another route to gather transformation data is to collect more specific data for different land use types, as Lindeijer *et al.* (1998) have done. Regionalisation implies adding regional information to land use intervention data, for instance by specifying the process type.

Finally, it may be queried whether the linear relationship between area, time and impact indicator is valid for all land use types. Including the area linearly is sometimes questioned because the area in question is used intentionally, with, in the case of forestry and agriculture, deliberate impacts on the environment (reducing biodiversity for the sake of productivity). This is different from impacts due to emissions. Also, biodiversity does not in fact vary linearly with area owing to the area-species relationships mentioned earlier. The assumption of linearity for non-critical areas may be adjusted in more refined modeling.

CONCLUSIONS

It is not currently feasible to select a satisfactory baseline method for loss of biodiversity and life support functions. There are too many flaws attached to all the methods currently available. Therefore, no baseline method is recommended. However, for LCA studies in which land use impacts may play a significant or even dominant role it is advised (particularly in detailed LCA studies) to use the methods of Lindeijer (1998), Köllner (2000) and, if possible, an operationalisation of the SETAC conceptual scheme for land use impacts.

Loss of biodiversity:

characterisation method	characterisation model	reference
baseline	–	–
alternative 1	based on a statistical measure of species density	Lindeijer *et al.* (1998)
alternative 2	based on a statistical measure of plant species density	Köllner (2000)
additional	–	–
variant	–	–

Loss of life support functions:

characterisation method	characterisation model	reference
baseline	–	–
alternative	based on Net Primary Production	Lindeijer *et al.* (1998)
additional	–	–
diverging	–	–

RESEARCH RECOMMENDATIONS

Short-term research

- It is highly recommended to start a project to arrive at one authorised set of methods for the inventory analysis and characterisation of the various impacts of land use.

Long-term research

- The difference between occupation impacts and transformation impacts in relation to the endpoints requires closer scrutiny. To this end irreversible versus temporary impacts should be distinguished. Impacts of different agricultural and silvicultural management systems can be expressed in terms of occupation impacts but also as long-term transformations. A distinction between occupation and transformation in data and Impact assessment is required. The acceptability of finally aggregating might be refined, but transformation data related to trends in land use changes is even more important, considering the probably greater environmental relevance of transformations relative to occupation.
- In general, further development of the indicator set for land use impacts is required. Sophistication should be improved through greater use of regionalised data and, if necessary, more indicators, supported by adequate inventory data, also for reference state indicators. The relationship between area of land use and impact indicators should also be studied in greater detail.
- Finally, other interventions leading to similar impacts (such as desiccation and intersection) may be operationalised using the same Impact assessment approach, and the distinction made between occupation and transformation impacts needs a framework for Interpretation.

4.3.4 Desiccation

TOPIC

Desiccation refers to a group of related environmental problems caused by water shortages due to groundwater extraction for industrial and potable water supply, enhanced drainage and water management (i.e. manipulation of the water table). This may lead to a lowered water table, reduced seepage, introduction of water from other areas and (consequently) changes in natural vegetation. The area of protection is the natural environment.

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)
In Heijungs *et al.*(1992) no indicator was recommended: "If the Inventory analysis yielded information about water use this could be totalled and used as a rough indicator for the issue of desiccation. Such a coarse approach, however, does not provide much more information about the actual issue…".

No useful methods have yet been developed for incorporating desiccation in LCA.

PROSPECTS

As mentioned earlier, RIZA and KIWA (NL) are currently examining the scope for incorporating a category indicator for desiccation in LCA. A follow-up project to operationalise desiccation using the method of either Lindeijer (1998) or Goedkoop & Spriensma (1999), or possibly both, is due to start by mid–2000. A research project on the topic is also under way in Australia.

CONCLUSIONS

No baseline method is recommended.

method status	characterisation method/factor	reference
baseline		–
alternative	–	–
additional	–	–
variant	–	–

RESEARCH RECOMMENDATIONS

No further research recommended.

4.3.5 Climate change

TOPIC

Climate change is defined here as the impact of human emissions on the radiative forcing (i.e. heat radiation absorption) of the atmosphere. This may in turn have adverse impacts on ecosystem health, human health and material welfare. Most of these emissions enhance radiative forcing, causing the temperature at the earth's surface to rise. This is popularly referred to as the 'greenhouse effect'. The areas of protection are human health, the natural environment and the man-made environment (see Figure 4.2.2).

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)

In Heijungs *et al.* (1992) *Global Warming Potentials* (GWPs) were used as characterisation factors to assess and aggregate the interventions for the impact category climate change (there termed "enhanced greenhouse effect"):

$$Climate\,change = \sum_i GWP_i \times m_i \qquad (4.3.5.1)$$

where m_i (kg) is the mass of substance i released in kg, GWP_i the Global Warming Potential of the substance and *Climate Change* the indicator result, which is expressed in kg CO_2-equivalents.

To compare the impacts of emissions of different greenhouse gases, each has been assigned a so-called Global Warming Potential (GWP) index, expressing the ratio between the increased infrared absorption due to the instantaneous emission of 1 kg of the substance and that due to an equal emission of carbon dioxide (CO_2), both integrated over time:

$$GWP_{T,i} = \frac{\int_0^T a_i c_i(t)dt}{\int_0^T a_{CO_2} c_{CO_2}(t)dt} \qquad (4.3.5.2)$$

with:

a_i	the radiative forcing per unit concentration increase of greenhouse gas i ($w \cdot m^{-2} \cdot kg^{-1}$),
$c_i(t)$	the concentration of greenhouse gas i at time t after the release ($kg \cdot m^{-3}$), and
T	the time over which integration is performed (yr).

The corresponding values for carbon dioxide are included in the denominator of the equation (Houghton *et al.*, 1991; Jäger & Ferguson, 1991). GWP is a measure of the potential contribution a substance to climate change and incorporates considerations of fate. It merely provides a rough indication of the potential climatic effects of such emissions, as these depend not only on integrated atmospheric heat absorption but also on its distribution over time.

This integration of the process of global warming involves a number of simplifications. In particular, GWPs depend on the time horizon T to which integration is performed. Longer horizons (100 and 500 years) are used to assess the cumulative effect of greenhouse gas emissions, while shorter horizons (20 and 50 years) provide an indication of short-term effects. The longer the time horizon the more unreliable GWPs become, as their value is determined by the background concentration of other components of the atmosphere. Although these concentrations are assumed to remain constant in time (Houghton *et al.*, 1992), they are likely to change. If assumptions about background concentrations change, GWPs also change. For example, the life span of substances eliminated by OH-radicals in the atmosphere may change significantly, depending on future changes in anthropogenic emissions of methane, carbon monoxide and nitrogen oxides (Houghton *et al.*, 1991; Jäger & Ferguson, 1991). Such changes in life span have a major, often disproportional, effect on the GWPs of all the substances concerned, as these are defined in relative terms.

The Intergovernmental Panel on Climate Change (IPCC) has compiled a list of 'provisional best estimates' for GWPs with time horizons of 20, 100 and 500 years, based on the expert judgement of scientists worldwide. This list of GWPs is periodically updated. The GWPs used in Heijungs *et al.* (1992) are based on the 1992 IPCC list (Houghton *et al.*, 1992).

The integration period to be applied in LCA calculations must be decided on by the practitioner and depends on the period over which the impacts are to be studied. A long horizon would appear to be preferable for the characterisation step of LCIA, as the aim of LCA is to assess all rather than just short-term effects. As stated, however, the longer the integration period, the more uncertainties are introduced into the model. Hence, Heijungs *et al.* (1992) recommended using all three IPCC time horizons. In practice this means that integration is first performed for 100 years, say, and the uncertainty margins in the result then determined by performing parallel calculations for the other two integration periods.

As mentioned, the IPCC's list of GWPs is periodically updated, for the first time in 1994 (Houghton *et al.*, 1994) and, for selected substances, again in 1996 (Houghton *et al.*, 1996). In this last update, a net GWP for ozone-depleting gases was discussed. While these gases absorb infrared radiation and thus increase radiative forcing, this is offset to some extent by a decrease in forcing due to the loss of stratospheric ozone. For these gases, therefore, a net GWP would be more appropriate than one based solely on direct impact. The net GWPs given in Houghton *et al.* (1996) show that while some ozone-depleting substances, like CFCs, still have a positive net GWP, for others, such as halons, this figure is negative. It is to be queried whether ozone layer depletion should be regarded as a 'positive' effect. Because negative impacts on the ozone layer are accounted for under the impact category 'depletion of stratospheric ozone', however, it seems appropriate to do so. At the same time, though, the net GWPs calculated to date are "subject to considerable quantitative uncertainties (at least 50%)" (Houghton *et al.*, 1996).

PROSPECTS

In the future, further progress is anticipated on Impact assessment modeling to the various endpoints, viz. damage to human health, damage to ecosystem health, etc. (*cf.* Figure 4.2.2); see Section 4.2 for a more extensive discussion of this subject).

CONCLUSIONS

Our conclusion is that the GWP approach, using the most recent indices published by the IPCC (Houghton *et al.*, 1994 and 1996), is currently the best available practice. Because of the uncertainties in net GWPs for ozone-depleting gases, these indices have not been included in the baseline method (criterion 12 for the selection of baseline characterisation methods). If these uncertainties can be narrowed down in further research, net GWPs should be used for ozone-depleting gases, as these are a more accurate reflection of our current understanding of relevant environmental mechanisms (criterion 1).
The GWPs for 100 years are recommended as the baseline characterisation method for climate change. The IPCC also provides GWPs for 20 and 500 years. Although 500 years is closer to eternity (criterion 7), we do not recommend using the GWPs for 500 years as the baseline, because of growing uncertainties in GWP with increasing time span (criterion 12).
The baseline characterisation method for climate change, using the characterisation factor GWP_{100}, is evaluated with respect to the (ISO-based) criteria adopted in this Guide in Table 4.3.5.1.

Table 4.3.5.1: Evaluation of the baseline characterisation method for climate change, using the characterisation factor GWP_{100}, with respect to the (ISO-based) criteria of Table 4.3.1.

	criterion	evaluation
1.	scientifically and technically valid	yes, calculation of the contributions of greenhouse gases to radiative forcing is based on well understood environmental processes
2.	environmentally relevant	yes, there is international agreement that climate change induced by radiative forcing will have environmental impacts such as sea level rise, destruction of coastal ecosystems, depressed crop yields, etc.
3.	internationally accepted	yes, supported by the Intergovernmental Panel on Climate Change (IPCC)
4.	value-choices and assumptions	yes, but agreed upon by an authoritative international body (IPCC).
5.	focal point in environmental mechanism	midpoint
6.	linearity	yes
7.	time span	100 years, not eternity
8.	fate, exposure/intake and effects	fate included, exposure/intake not relevant, effects included in terms of effects on radiative forcing
9.	less is better	yes, no threshold

	criterion	evaluation
10.	time- and location-independent	yes
11.	operational	yes
12.	uncertainty margins	known and accepted by IPCC

The following method is included in this Guide as an option for sensitivity analyses:
The GWPs for 20 and 500 years can be used to explore the consequences of adopting different time horizons.

Recommendations for extended LCAs:
– For several ozone-depleting gases, the upper and lower limits of the uncertainty range of net GWPs may be useful in an extended LCA.

method status	characterisation method/factor	reference
baseline	GWP_{100}	Houghton *et al.*, 1994, 1996
alternative 1	GWP_{20}	Houghton *et al.*, 1994, 1996
alternative 2	GWP_{500}	Houghton *et al.*, 1994, 1996
alternative 3	upper limit of net GWP	Houghton *et al.*, 1996
alternative 4	lower limit of net GWP	Houghton *et al.*, 1996
additional	–	–
variant	–	–

RESEARCH RECOMMENDATIONS
Given the ongoing work of the IPCC, no further research is recommended.

4.3.6 Stratospheric ozone depletion

TOPIC
Stratospheric ozone depletion refers to the thinning of the stratospheric ozone layer as a result of anthropogenic emissions. This causes a greater fraction of solar UV-B radiation to reach the earth's surface, with potentially harmful impacts on human health, animal health, terrestrial and aquatic ecosystems, biochemical cycles and materials (UNEP, 1998). Stratospheric ozone depletion thus impinges on all four areas of protection: human health, the natural environment, the man-made environment and natural resources (see Figure 4.2.2).

DEVELOPMENTS IN THE LAST DECADE
The concept of Ozone Depletion Potential (ODP) was introduced by Wuebbles (1988). The ODP of a substance is defined as follows:

$$ODP_i = \frac{d[O_3]_i}{d[O_3]_{CFC-11}} \qquad (4.3.6.2)$$

Heijungs *et al.* (1992)
In Heijungs *et al.* (1992) *Ozone Depletion Potentials* (ODPs) were used as a characterisation factor to assess and aggregate the interventions for the impact category stratospheric ozone depletion:

$$Ozone\ depletion = \sum_i ODP_i \times m_i \qquad (4.3.6.1)$$

where m_i (kg) is the mass of substance *i* released, ODP_i the Ozone Depletion Potential of the substance and *Ozone Depletion* the indicator result, which is expressed in kg CFC–11-equivalents.

with:

$d[O_3]_i$ represents the change in the stratospheric ozone column i the equilibrium state due to annual emissions of substance *i* (flux in kg.yr^{-1}), and

$d[O_3]_{CFC-11}$ the change in this column in the equilibrium state due to annual emissions of CFC–11.

It can be shown that an ODP based on an emission flux (kg.yr^{-1}) also provides a good indication of the relative changes in the ozone column due to an instantaneous emission (kg) to the atmosphere (WMO, 1989). Although the ODP concept resembles that of GWP, there is a major difference: ODPs are calculated for a steady state, GWPs for several different time horizons.

The *World Meteorological Organisation* (WMO) has compiled a list of 'best estimates' for ODPs, first published in 1992 (WMO, 1992). The report was compiled by the *Scientific Assessment Panel*, comprising the relevant authorities in the field, and it is therefore reasonable to assume that the results enjoy wide international support. The ODPs used in Heijungs *et al.* (1992) are based on this original list, which the WMO updated in 1995 (WMO, 1995) and again in 1999 (WMO, 1999).

These ODPs are steady-state ODPs based on a model. They describe the integrated impact of an emission of a substance on the ozone layer compared with CFC–11. These model-derived ODPs are recommended by Nichols *et al.* (1996) and by Hauschild & Wenzel (1998) for situations in which the time span of interest is eternity. In other cases Nichols *et al.* (1996) and WMO (1995) recommend using the time-dependent ODPs given by Solomon & Albritton (1992). These time-dependent ODPs are based on an empirical approach, viz. on measurements on the lower layers of the stratosphere. Hauschild & Wenzel (1998) also suggest that the time-dependent ODPs might be better for LCAs with a shorter horizon. For the time being, however, they opt to use the steady-state ODPs in their EDIP methodology.

Lindfors (1996) has evaluated the major impact categories for LCA within the framework of the EU ecolabeling programme. He draws a different conclusion from Nichols *et al.*(1996) and Hauschild & Wenzel (1998), stating that the category of stratospheric ozone depletion will become less important in the future because of the ongoing phase-out of ozone-depleting chemicals. He recommends not using ODPs, because of all the uncertainties involved (mainly data gaps in the inventory). Instead he proposes classifying and/or flagging emissions of ozone-depleting chemicals or use of ozone-depleting chemicals in products or technologies, with no further characterisation. It should be borne in mind, however, that this is specifically for the underpinning of ecolabeling criteria. For LCIA in general, using the best available method to characterise stratospheric ozone depletion seems to be a better approach than using no method at all. It is questionable, furthermore, whether emissions of ozone-depleting are set to decline at the global level, l*et al*one stratospheric concentrations.

PROSPECTS

In the future, further progress is anticipated on Impact assessment modeling to the various endpoints, viz. damage to human health, damage to ecosystem health, etc. (*cf.* Figure 4.2.2); see Section 4.2 for a more extensive discussion of this subject).

CONCLUSIONS

As a default for both simplified and detailed LCAs it is recommended to use the most recent steady-state ODPs published by the WMO (1999), augmented where necessary by ODPs from the original 1992 list (WMO, 1992). These values are internationally accepted (criterion 3 for selection of baseline characterisation method) and a steady-state ODP approximates an ODP for T = ∞ (criterion 7).

The baseline characterisation method for stratospheric ozone depletion, using the characterisation factor ODP_∞, is evaluated with respect to the (ISO-based) criteria adopted in this Guide in Table 4.3.6.1.

Table 4.3.6.1: Evaluation of the baseline characterisation method for stratospheric ozone depletion, using the characterisation factor ODP_∞, with respect to the (ISO-based) criteria of Table 4.3.1.

	criterion	evaluation
1.	scientifically and technically valid	yes, calculation of stratospheric ozone depletion is based on well understood environmental processes

	criterion	evaluation
2.	environmentally relevant	yes, there is international agreement that stratospheric ozone depletion results in an increase of UV-B intensity, causing a variety of radiation effects on humans, algae, arctic flora, crops, etc.
3.	internationally accepted	yes, supported by the World Meteorological Organisation (WMO) and the United Nations
4.	value-choices and assumptions	yes, but agreed on by an authoritative international body (WMO).
5	focal point	midpoint
6.	linearity	yes
7.	time span	eternity
8.	fate, exposure/intake and effects	fate included, exposure/intake not relevant, effects included in terms of effects on stratospheric ozone layer
9.	less is better	yes, no threshold
10.	time- and location-independent	yes
11.	operational	yes
12.	uncertainty margins	known and accepted by WMO

The following method is included in this Guide as an option for sensitivity analyses:
The time-dependent ODPs of Solomon & Albritton (1992) can be used to explore the consequences of adopting different time horizons.

No additional methods are recommended for extended LCAs.

method status	characterisation method/factor	reference
baseline	ODP_{∞}	WMO, 1992; WMO, 1995; WMO ,1999
alternative 1	ODP_{5}	Solomon & Albritton, 1992
alternative 2	ODP_{10}	Solomon & Albritton, 1992
alternative 3	ODP_{15}	Solomon & Albritton, 1992
alternative 4	ODP_{20}	Solomon & Albritton, 1992
alternative 5	ODP_{25}	Solomon & Albritton, 1992
alternative 6	ODP_{30}	Solomon & Albritton, 1992
alternative 7	ODP_{40}	Solomon & Albritton, 1992
additional	–	–
variant	–	–

RESEARCH RECOMMENDATIONS

Long-term research:

- Research is recommended to identify the differences and similarities between the methods used to derive ODPs and GWPs.

4.3.7 Human toxicity

TOPIC

This impact category covers the impacts on human health of toxic substances present in the environment. The health risks of exposure in the workplace are also sometimes included in LCA (see, for example, Lindfors *et al.*, 1995c; Hauschild & Wenzel, 1998; Schmidt & Brunn Rasmussen, 1999). These latter risks are often included in a wider impact category encompassing more than exposure to toxic substances (e.g. accidents at work). Schmidt & Brunn Rasmussen (1999) describe a useful method for including the working environment in LCA, based on a database developed by EDIP in which workplace impacts per kilogram of produced goods are listed for a number of economic activities. However, the scope of the present project precluded assessment of whether this method could be adapted and incorporated in this Guide. No further consideration is therefore given here to the impacts of

exposure to toxic substances in the workplace. The area of protection for this impact category is human health (see Figure 4.2.2).

A variety of characterisation methods have been developed for the impact category human toxicity providing characterisation factors that are generally referred to as 'human toxicity potentials' (HTPs). As described by Heijungs & Wegener Sleeswijk (1999) the general formula for calculating the HTP embraces three independent dimensions: fate, exposure/intake and effect. Here, we add a fourth dimension to account for transfer:

$$HTP_{i,ecomp} = \sum_{fcomp} \sum_{r} F_{i,ecomp,fcomp} \times T_{i,fcomp,r} \times I_r \times E_{i,r} \qquad (4.3.7.1)$$

with:

$HTP_{i,ecomp}$ the Human Toxicity Potential, the characterisation factor for the human toxicity of substance *i* emitted to emission compartment *ecomp*. In some methods the contributions via exposure routes *r* are not summed, yielding several HTPs.

$F_{i,ecomp,fcomp}$ a 'fate factor', representing intermedia transport of substance *i* from emission compartment *ecomp* to final (sub)compartment *fcomp*, and degradation within compartment *ecomp*; in some methods intermedia transport is indicated separately by $f_{i,ecomp,fcomp}$ and (bio)degradation by BIO_i;

$T_{i,fcomp,r}$ the transfer factor, the fraction of substance i transferrred from *fcomp* to exposure route *r*, i.e. air, drinking water, fish, plants, meat, milk, etc.;

I_r an 'intake factor', representing human intake via exposure route *r*; thus, a function of daily intake of air, drinking water, fish, etc.;

$E_{i,r}$ an 'effect factor', representing the toxic effect of intake of substance *i*' via exposure route *r*.

The HTP is often defined relative to a reference substance. As a formula:

$$HTP_{i,ecomp} = \frac{\sum_{fcomp} \sum_{r} F_{i,ecomp,fcomp} \times T_{i,fcomp,r} \times I_r \times E_{i,r}}{\sum_{fcomp} \sum_{r} F_{refi,refecomp,fcomp} \times T_{refi,fcomp,r} \times I_r \times E_{refi,r}} \qquad (4.3.7.2)$$

with the symbols similar to the above[1]. The choice of the reference substance is arbitrary.

This general description of the HTP-formula and the associated terminology provides an initial handle for describing and discussing the various characterisation methods developed for the impact category human toxicity. Thus, the terms used in this Guide to describe these methods may differ from those employed by the authors in question.

[1] The HTP using a reference substance is dimensionless or has the unit 'kg reference substance·kg $^{-1}$ of substance *i*''. In this section and the following it is considered as a dimensionless quantity, although in Part 2b it is precisely quantified.

A second point of terminology concerns the environmental 'compartments' or 'media', used for the three basic subdivisions air, water and soil. In this Guide 'compartment' is used as the preferred term, except in the standard collocations 'multimedia' and 'intermedia'. 'Subcompartments' are subdivisions of air, water or soil that are in chemical equilibrium with the compartment of which they are a part. Examples of subcompartments are aerosols, suspended matter and pore water in soils. Compartments or subcompartments through which humans are exposed to a toxic substance are referred to as exposure routes (see textbox for some further details).

General structure of models for toxicity assessment

The impact categories relating to toxicity-oriented problems can be described in terms of the main aspects covered (see also description under Topic). These aspects are:

1. Fate. The residence time of a chemical in a particular environmental compartment depends on degradation mechanisms and transport processes, e.g. from air to soil by rain, from water to air by evaporation, from soil to water by run-off.
2. Transfer. The fraction of a substance transferred from a given compartment to an exposure route, i.e. air, drinking water, fish, plants, meat, milk.
3. Exposure/intake. The intake of a given chemical by an organism depends on its food pattern, water intake and respiratory volume.
4. Effect. There is wide variation in the hazards posed by chemicals; dioxins, for example, are more toxic than nitrates.

Fate, transfer and exposure/intake are often modeled together. A number of compartments and sub-compartments are distinguished, e.g. air, soil, freshwater, marine waters and sediment. Environmental processes like rainfall, degradation, sedimentation and immobilisation (e.g. by burial in deeper sediments) are captured in model equations, which are then extended via exposure routes (air, drinking water, crops, meat, milk, fish) to target organisms, including humans, or to target ecosystems (terrestrial, freshwater, marine). As noted in Section 1.2.3.3 indirect exposure to humans via all manner of foodstuffs is a very significant exposure route for many substances. Exposure by this route depends on foodstuff consumption as such, but also on bioconcentration and biomagnification in all these foodstuffs, which are derived directly or indirectly from plants, including meat and dairy. Because of the enormous variety of foodstuffs we consume, the soil route is by far the most complex route for transfer modeling. Most fate, transfer and exposure/intake models are steady-state models, i.e. they calculate a concentration or intake level due to a constant emission rate. Dynamic models, on the other hand, yield a pattern in time, as the result of a constant, varying or pulse emission input. Time-integration then condenses such a pattern into a single result. Steady-state models are also applied to assess emission pulses. It can be shown that most steady-state models based on an emission flux ($kg.yr^{-1}$) also provide a good indication of the relative changes in fate and exposure/intake due to an emission flux (kg) (Guinée *et al.*, 1996). Most fate models assume homogenous mixing, but occasionally lagrangian or gaussian models are used to calculate concentration gradients. Space-integration may then also be required. Homogenous box-models may allow for discrete regional differences as well, e.g. at the scale of continents, climatic zones or countries. Space-integration in that case amounts to volume-weighted addition.

The effect measure is often based on toxicologically-based yardsticks such as EC50, defined as the concentration at which 50% of the target organisms shows an effect. Extrapolation or safety factors may be applied to convert the results from the laboratory to the field, from rat to man, from single species to ecosystem, etc, yielding NOECs (No Observed Effect concentration), ADIs (Acceptable Daily Intakes), TDIs (Tolerable Daily Intakes), PNECs (Predicted No-Effect Concentrations), MTCs (Maximum Tolerable Concentrations), NOAELs (No Observed Adverse Effect Levels) and so on. The terms 'acceptable', 'tolerable' and 'adverse' imply that these measures are mainly based on toxicological (rather than economic, political, etc.) considerations.

DEVELOPMENTS IN THE LAST DECADE

Human toxicological impact, together with ecotoxicological impact, is the impact category for which fate and especially degradation and intermedia transport are most important. Organic substances, in particular, are generally degraded to yield other compounds that are less toxic than the substance originally emitted (although the opposite may also be true). Furthermore, substances do not generally remain in the environmental compartment into which they are emitted, but tend to spread to other (sub-)compartments, where they may do more damage. A volatile toxic substance discharged into waterways will evaporate largely to the atmosphere, for example, where it may expose humans to (severe) risk via the respiratory route.

Many authors therefore stress that it is essential to properly incorporate degradation and intermedia transport in LCA models of human toxicological impact (e.g. Heijungs *et al.*, 1992; Lindfors *et al.*, 1995c; Udo de Haes *et al*, 1996; Guinée *et al.*, 1996; Jolliet & Crettaz, 1997; Hauschild & Wenzel, 1998).

In line with these recommendations, we here consider only methods that include degradation and intermedia transport, which supersede the provisional method developed by Heijungs *et al.* (1992)[1]. (See text box.) Several such methods are described in the literature, falling roughly into three groups:

1. methods in which degradation and intermedia transport is based on simple rules of thumb (e.g. Hauschild & Wenzel, 1998)
2. methods in which degradation and intermedia transport is based on models (e.g. Guinée *et al.*, 1996; Huijbregts, 1999a; Hertwich, 1999)

Heijungs *et al.* (1992)

Heijungs *et al.* (1992) defined separate characterisation factors for emissions of toxic substances to the environmental compartments air, water and soil:

$$HCA = \frac{VI_a \times W}{V_a \times ADI} \qquad (4.3.7.3)$$

$$HCW = \frac{VI_w \times W}{V_w \times ADI} \qquad (4.3.7.4)$$

$$HCS = \frac{M \times W \times N}{V_S \times C\,value} \qquad (4.3.7.5)$$

where HCA, HCW and HCS are the characterisation factors for human toxicological impacts resulting from emissions to air, water and soil, respectively (kg body weight·kg^{-1} substance). VI_a and VI_w are the daily intakes of air and water per person (= 20m^3 air·day^{-1}·person^{-1} and 2 l water·day^{-1}·person^{-1}). W is the world population (= 5·10^9 persons). V_a, V_w and V_s are the volumes of air, water and soil in the world (=3·10^{18} m^3 air, 3.5·10^{18} l water and 2.7·10^{16} kg dry soil). ADI is the Acceptable Daily Intake: for substances with a threshold value (i.e. an environmental concentration or intake value below which no harmful effects have been observed in humans, plants or animals) it is the daily intake that can be sustained life-long without adverse effects; for substances with no such threshold, it is the daily intake resulting in a risk of 1 extra case of cancer per 1000 life-long exposures (VROM, 1989). ADI is expressed as kg substance·day^{-1}·kg^{-1} body weight. N is the uncertainty factor for the ADI and Cvalue is a former Dutch standard for soil (kg substance ·kg^{-1} soil).

The indicator results for these media can be added without weighting to provide a single, medium-independent indicator result for human toxicity:

$$Human\ toxicity = \sum_i \left(\left(HCA_i \times m_{a,i}\right) + \left(HCW_i \times m_{w,i}\right) + \left(HCS_i \times m_{s,i}\right) \right) \qquad (4.3.7.6)$$

where $m_{x,i}$ (kg) are the emissions of substance i to air, water and soil.

Heijungs *et al.* stress that these characterisation factors for human toxicity should be considered as no more than indicative until such time as a better method is developed.

[1] Heijungs *et al.* stressed in 1992, already, that fate should be included; at the time, however, this could be done only provisionally.

3. methods in which degradation and intermedia transport is based on models and empirical relations (e.g. Jolliet & Crettaz, 1997).

Below, these methods are discussed in terms of a common notation as described above. For instance, F is used consistently to denote a fate factor, although a variety of symbols and terms are encountered in the literature. Moreover, the precise operationalisation and, consequently, the units used for broadly similar variables may differ from method to method.

1. Methods based on simple rules of thumb

Hauschild & Wenzel (1998) have developed characterisation factors based on rules of thumb to model degradation and intermedia transport:

$$HTP_{i,ecomp,r} = \sum_{fcomp} f_{i,ecomp,fcomp} \times BIO_i \times T_{i,fcomp,r} \times I_r \times E_{i,r} \qquad (4.3.7.7)$$

with:

$HTP_{i,ecomp,r}$[1] the Human Toxicity Potential, the characterisation factor for the human toxicity of substance *i* emitted to emission compartment *ecomp* and leading to exposure via route *r* (e.g. fish or milk; see Figure 4.3.7.1). The emission compartments considered are air, water and soil, resulting in nine different HTPs per substance. Groundwater has been suggested as a fourth compartment, but no characterisation factors have yet been developed;

$f_{i,ecomp,fcomp}$ the intermedia transport factor, the fraction of substance *i* emitted to emission compartment *ecomp* that reaches final compartment *fcomp* as a result of environmental transport. This factor is based on simple rules of thumb rather than on a full fate model. Moreover, it is not a continuous variable but assumes only a limited number of values such as 0.2 and 1;

BIO_i the biodegradability factor, to be chosen from one of three classes depending on the substance *i* involved;

$T_{i,fcomp,r}$ the transfer factor, the fraction of substance *i* transferrred from *fcomp* to exposure route r, i.e. air, fish, plants, animals, etc. (see Figure 4.3.7.1). In the case of soil the HTP includes only those exposure routes making the greatest contribution (i.e. with the highest $I_{i,r} \cdot T_{i,fcomp,r}$);

I_r the intake factor, the fraction of substance *i* taken up per kg body weight per day via exposure route r. There is a specific intake factor for each of the seven sub-compartments identified;

$E_{i,r}$ the effect factor, representing the human-toxic impact of substance *i* via exposure route r; depending on the exposure route concerned, it is either the reciprocal of the Acceptable Daily Intake or that of the atmospheric concentration not anticipated to have toxic effects after life-long inhalation.

[1] Hauschild & Wenzel use a different term for HTP, viz. 'Equivalency Factor'. Moreover, Hauschild & Wenzel use the term 'potential' for the associated indicator results, whereas in this Guide the term is used for the characterisation factor. Finally, *r* is used here to indicate exposure routes such as 'air', 'fish' and so on (see Figure 4.3.7.1 and Figure 4.3.7.2), whereas Hauschild & Wenzel indicate these by abbreviations based on media: a = air, viz. respiration; w = water, viz. fish; s = soil, viz. crops (plants in Figure 4.3.7.1), cattle meat (animals in Figure 4.3.7.1) and dairy products (milk in Figure 4.3.7.1).

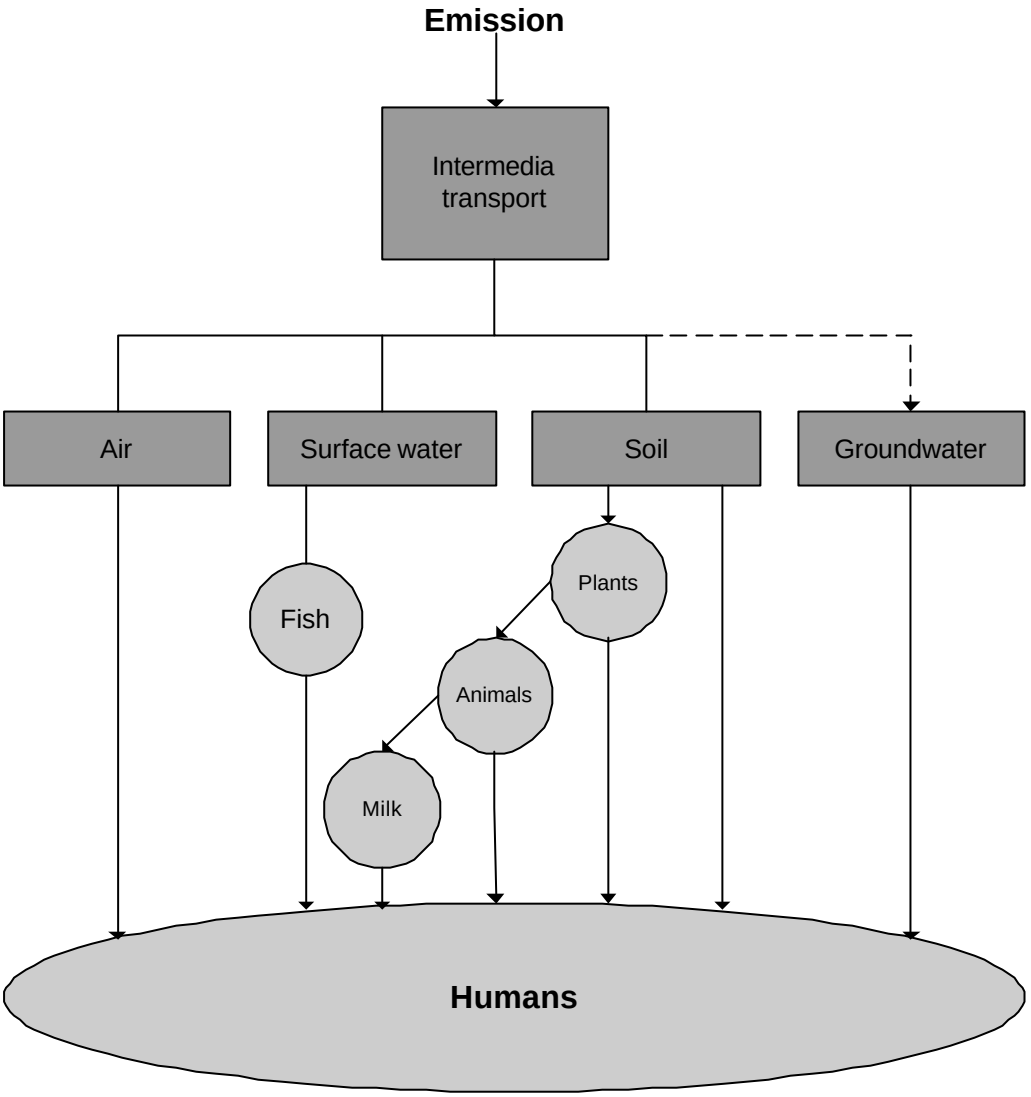


Figure 4.3.7.1: The compartments and seven exposure routes considered in the human toxicity approach of Hauschild & Wenzel (slightly adapted from figure presented by Wenzel *et al.*, 1997).

The method of Hauschild & Wenzel (1998) yields three separate indicator results for human toxicity, one for each of the three principal exposure routes (air; fish; and soil, plants, animals and milk):

$$human\ toxicity_r = \sum_i \sum_{ecomp} m_{i,ecomp} \times HTP_{i,ecomp,r} \qquad (4.3.7.8)$$

where $HTP_{i,ecomp,r}$ is as defined above and $m_{i,ecomp}$ is the emission of substance *i* to compartment *ecomp*. These separate results are not aggregated to a single indicator result.

Overall, it can be concluded that the method of Hauschild & Wenzel (1998) models degradation and intermedia transport only very rudimentarily.

2. Model-based methods

Guinée *et al.* (1996) have developed characterisation factors for human toxicity including degradation and intermedia transport using the Uniform System for the Evaluation of Substances model, USES 1.0 (RIVM *et al.*, 1994), which incorporates the multimedia model Simplebox 1.0 (Van de Meent, 1993) as a separate module. Simplebox calculates the Predicted Environmental Concentration (PEC) in four environmental compartments, represented as 'boxes': air, water, agricultural soil and industrial soil (i.e. soil used for agricultural and industrial purposes) due to a constant emission flux to any of these four

compartments. For human toxicity, six exposure routes are assessed, viz. air, fish, drinking water, crops, cattle meat and milk (see Figure 4.3.7.2).

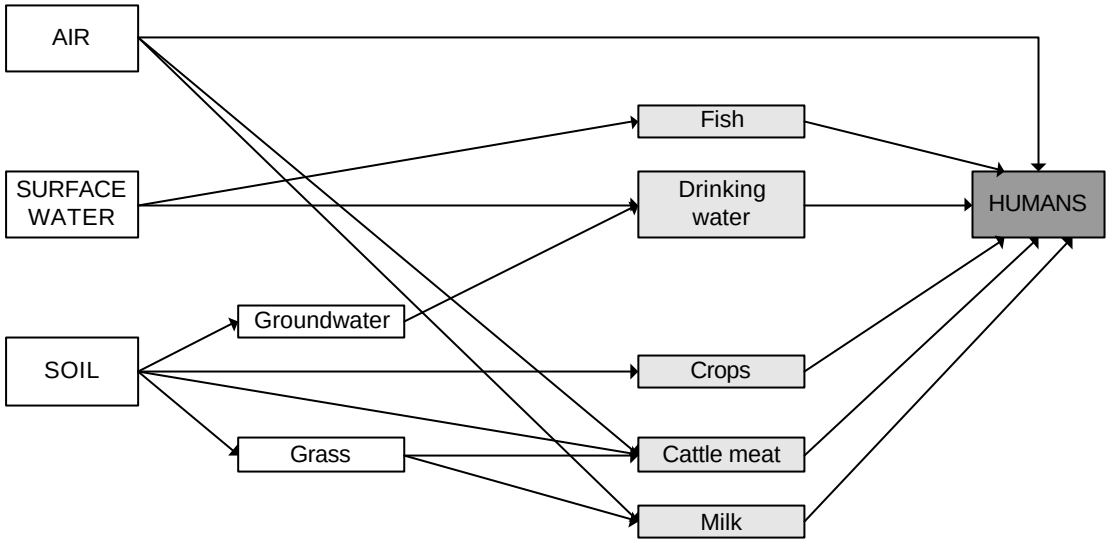


Figure 4.3.7.2: Human exposure routes identified in USES 1.0 (source: Guinée *et al.*, 1996).

Although exposure route and intake are still modeled in a fairly rudimentary fashion, using similar methods to Hauschild & Wenzel (1998), Simplebox allows substance fate to be modeled rather more realistically and comprehensively, including both degradation and immobilisation and making use of continuous functions that depend on the chemical of interest rather than the constant factors *f* and *BIO*.

A second major difference in the method of Guinée *et al.* is that the daily intakes accruing via the respective exposure routes are summed to yield a total Predicted Daily Intake (PDI)[1], which is then divided by the Acceptable Daily Intake (ADI), expressed here once more as its reciprocal (E_i). Finally, the score is divided by the score for an emission, to air, of a reference substance: 1,4 dichlorobenzene:

$$HTP_{i,ecomp} = \frac{PDI_{i,ecomp} \times E_i}{PDI_{air,1,4-dichlorobenzene} \times E_{1,4-dichlorobenzene}} \qquad (4.3.7.9)$$

with:

$HTP_{i,ecomp}$ the Human Toxicity Potential due to emission of 1000 kg of substance *i* per day (flux) to emission compartment ecom. The compartments considered are air, water and agricultural and industrial soil, resulting in four different HTPs per substance;

$PDI_{i,ecomp}$ the Predicted Daily Intake of substance *i* emitted to emission compartment *ecomp*; note that $PDI_{i,ecomp}$ does not differ for different exposure routes since in this method E_i does not differ among routes;

E_i the effect factor, representing the human-toxic impact of substance *i*, here the reciprocal of the ADI of the substance;

$PDI_{air,\ 1,4\text{-diochlorobenzene}}$ the Predicted Daily Intake resulting from the emission of 1000 kg of 1,4-dichlorobenzene per day to air;

$E_{1,4\text{-dichlorobenzene}}$ the effect factor for 1,4-dichlorobenzene, representing the human-toxic impact of 1,4-dichlorobenzene, here the reciprocal of the ADI of 1,4-dichlorobenzene.

It can be shown that an HTP based on an emission flux ($kg.yr^{-1}$) also provides a good indication of the relative human-toxic impact of an instantaneous emission (kg) (Guinée *et al.*, 1996).

The method of Guinée *et al.* (1996) results in a single indicator result for human toxicity:

[1] $PDI_{i,ecomp}$ is this a combination of $F_{i,ecomp,fcomp}$, $T_{i,fcomp,r}$ and I_r.

$$human\ \ toxicity\ = \sum_{i} \sum_{ecomp} m_{i,ecomp} \times HTP_{i,ecomp} \qquad (4.3.7.10)$$

where $HTP_{i,ecomp}$ and $m_{i,ecomp}$ are as defined above.
Equation (4.3.7.10) can also be written as:

$$human\ \ toxicity\ = \sum_{i} \sum_{ecomp} m_{i,ecomp} \times \left[HTP_{i,ecomp\ ,r1} + HTP_{i,ecomp\ ,r2} + ...\right] \qquad (4.3.7.11)$$

where r1 is exposure route 1, e.g. air, r2 exposure route 2, e.g. fish, and so on. Equation (4.3.7.11) clearly shows the difference from the method of Hauschild & Wenzel (1998), which yields separate indicator results for each of the exposure routes distinguished rather than their sum, here.

Huijbregts (1999a) used a new version of the USES model, USES 2.0[1] (RIVM *et al.*, 1998), and modified it to calculate new characterisation factors for human toxicity (as well as for aquatic, sediment and terrestrial ecotoxicity), using the same basic method as Guinée *et al.* (1996). The USES-LCA model thereby created (largely based on USES 2.0) improves on USES 1.0 in four main ways. In the first place, the fate of substances can now be modeled at the global level. USES 2.0 and USES-LCA have five spatial scales: 'regional', 'continental' and 'global', the last tripartite to reflect the arctic, temperate and tropical climate zones of the Northern hemisphere. The regional and continental scales each comprise six compartments: air, freshwater, seawater, natural soil, agricultural soil and industrial soil. The three climate zones of the global scale each comprise three compartments: air, (sea)water and soil. The global scale is modeled as a closed system with no transport out of the system (i.e. into space); emitted substances cannot therefore leave the system, as was the case in USES 1.0. Second, USES-LCA takes into account the temperature dependence of physico-chemical properties. Third, the variation of these properties with soil depth is also modeled. Finally, the continental-scale model now has separate freshwater and seawater compartments.

Given the differences between the USES-LCA and USES 1.0 models, Huijbregts (1999a) also differed from Guinée *et al.* (1996) in a number of choices. The principal of these derives from the fact that USES-LCA is a nested model, with five scale, as described above. USES 1.0, in contrast, had only a single, continental scale[2]. To calculate a single characterisation factor for each emission compartment, Huijbregts therefore aggregated the four factors calculated at the global and continental scales on a population basis: the larger the exposed population, the greater the weight of the associated factor:

$$HTP_{i,ecomp} = \frac{\sum_{r} \sum_{s} PDI_{i,ecomp\ ,r,s} \times E_{i,r} \times N_s}{\sum_{r} \sum_{s} PDI_{1,4-dichlorobenzene,air,r,s} \times E_{1,4-dichlorobenzene,r} \times N_s} \qquad (4.3.7.12)$$

with:

$HTP_{i,ecomp}$	the Human Toxicity Potential of substance *i* emitted to emission compartment *ecomp* (dimensionless);
N_s	the population density at scale s;
$PDI_{i,ecomp,r,s}$	the Predicted Daily Intake via exposure route *r* at scale s for substance *i* emitted to emission compartment *ecomp* (day^{-1});
$E_{i,r}$	the effect factor, representing the human-toxic impact of substance *i,* here the Acceptable Daily Intake via exposure route *r* (inhalation or ingestion) (day).

Although USES-LCA models fate and exposure routes more realistically and comprehensively than earlier methods, using continuous variables, the toxicity potentials thus calculated still embody areas of major uncertainty. The sensitivity of the USES-LCA model to (value) choices it embodies also needs to be carefully assessed. . The main value choices concern the temporal and spatial horizons adopted to

[1] USES 2.0 is an adapted version of EUSES, in turn an upgraded and updated European version of USES 1.0. .

[2] Although both USES 1.0 and 2.0 permit calculation on a local and regional scale, these scales were not taken into account in calculating characterisation factors.

calculate HTP. In the case of the impact category 'climate change', the value of the characterisation factor GWP may vary by more than an order of magnitude, depending on the time horizon chosen (Houghton *et al.*, 1996). In the case of toxicity potentials, an infinite time horizon has generally been adopted (Guinée *et al.*, 1996; Hertwich *et al.*, 1998 & 1999; Huijbregts 1999a; Huijbregts, 2000). In Impact assessment this approach may obscure the potential shorter-term impacts of product systems, however. With respect to spatial horizon, the basic choice concerns whether or not HTP includes the potential impacts exported from the continental to the global scale. As mentioned, Huijbregts (2000) includes the latter using scale-specific, population-based weighting factors. However, including global-scale impacts may obscure potential impacts at the continental scale.
Huijbregts therefore ran a number of scenarios to assess the influence of these choices (Huijbregts, 2000). Toxicity potentials were calculated for horizons of 20, 100 and 500 years by integrating the amount of a substance present in compartment *fcomp* after an emission pulse released to compartment *ecomp* over the respective periods and compared with the value previously obtained by integration to infinity (Figure 4.3.7.3 and Figure 4.3.7.4). These three horizons are the same as those used to calculate Global Warming Potentials (Houghton *et al.*, 1996) and appear to provide a practicable range for policy applications.

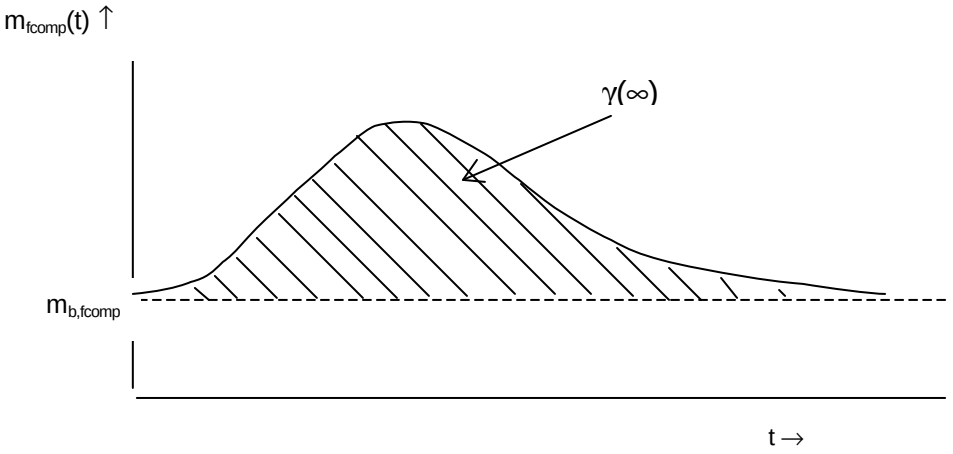


Figure 4.3.7.3: Schematic representation of integrated amount γ to an infinite time horizon in compartment *fcomp* after an emission pulse m released to compartment *ecomp* at t = 0, superimposed on a background level $m_{b,fcomp}$ (derived from Heijungs, 1997a).

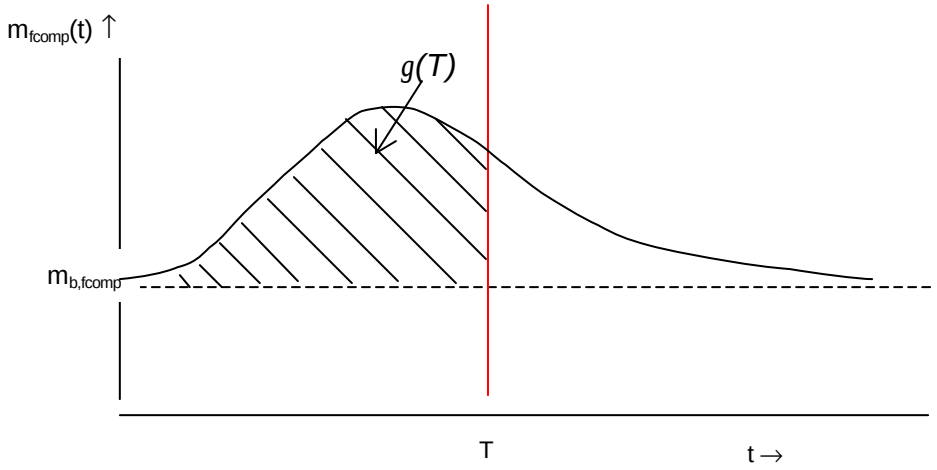


Figure 4.3.7.4: Schematic representation of integrated amount $\gamma(T)$ to time horizon *T* in compartment *fcomp* after an emission pulse Δm released to compartment *ecomp* at t = 0, superimposed on a background level $m_{b,fcomp}$.

The sensitivity of the model to the choice of spatial horizon was assessed by comparing toxicity potentials calculated with and without inclusion of the global scale. To this end Huijbregts (2000) aggregated potential impacts at the continental scale and each of the three zones of the global scale using scale-specific weighting factors. Potential impacts in the marine aquatic compartments were aggregated on the basis of compartment volume, impacts in the marine sediment and terrestrial compartments on the basis of compartment mass. For human toxicity, the human population at the scale level in question was used as a weighting factor. For the impact categories associated with the freshwater aquatic and sediment compartments no weighting factors were required, as these compartments are identified at the continental scale only. In calculating toxicity potentials, potential global-scale impacts can be excluded by assigning a zero value to the weighting factors for the contributions of arctic, temperate and tropical zones to the impact categories involved. The results of the time horizon scenario analyses (see Part 2b, Section 4.3.8, and see Huijbregts, 2000) show differences of up to 6.5 orders of magnitude for the toxicity potentials of the metals studied, while for organic chemicals these differences remain within half an order of magnitude. In terms of the LCA result for the impact category human toxicity, this means that the longer the time horizon, the more dominant the very persistent heavy metals become. Scenario analyses addressing the extent to which inclusion of global-scale impacts obscures potential continental-scale impacts indicate differences in the toxicity potentials of metals and volatile persistent halogenated organics of up to 2.3 orders of magnitude.
For a more extensive discussion of these results, we refer to Huijbregts (2000).
The use of time horizon-specific HTPs in LCAs is relatively straightforward. The indicator result for human toxicity and a specified time horizon can be calculated using the formula:

$$human\ \ toxicity_t = \sum_i \sum_{ecomp} m_{i,ecomp} \times HTP_{i,ecomp,t} \qquad (4.3.7.13)$$

with:

human toxicity$_t$	the indicator result for human toxicity for time horizon *t* (kg);
HTP$_{i,ecomp,t}$	the Human Toxicity Potential of substance *i* emitted to emission compartment *ecomp* for time horizon *t* (dimensionless);
m$_{i,ecomp}$	the emission of substance *i* to compartment *ecomp* (kg).

As metals and volatile persistent halogenated organics may be responsible for a substantial share of the potential human-toxic impact of product systems, the (value) choice of temporal and spatial horizon is particulartly important in LCA Impact assessment of toxic substances.

Besides embodying value choices, USES-LCA is also characterised by a number of major modeling uncertainties (Huijbregts *et al.*, 2000b; Ragas *et al.*, 1999). In this respect its modeling of the fate of metals is particularly weak. In the first place, box models such as USES-LCA include no spatial differentiation of fate, exposure/intake or effect parameters. Although in the case of organic chemicals spatial variability may not be very important in fate and effect assessment compared with the influence of parameter uncertainties (Hertwich *et al.*, 1999), this may not be true of metals. Intermedia transport of metals is highly dependent on environmental conditions, leading to major potential variation in residence time, particularly in the soil compartment (De Vries & Bakker, 1998). For metals, then, adoption of spatially differentiated models (e.g. Klepper & Den Hollander, 1999; Van den Hout *et al.*, 1999; Stolwijk *et al.*, 1998) will lead to improved assessment of fate, exposure/intake and ultimate effects. Such models would also permit inclusion of site-dependent processes that are currently lacking in USES-LCA, such as slow conversion of reversibly adsorbed heavy metals into forms irreversibly adsorbed to the soil matrix (Harmsen, 1992; De Vries & Bakker, 1998) and uptake by organisms (Peijnenburg *et al.*, 1997, 1999). Further research in the LCA context is recommended here.
The second aspect relates to the fate of geochemically reactive metals such as beryllium in the marine environment. Goldberg (1965) reports an oceanic residence time of Be three orders of magnitude lower than calculated by USES-LCA (Huijbregts, 2000). This is because Be-ions are anticipated to be rapidly hydrolysed by the pH of seawater and incorporated into minerals such as ferro-mangenese nodules (Goldberg, 1965; Riley, 1971). This removal mechanism is not included in USES-LCA. For other metals such as copper, zinc, lead, cadmium and vanadium, too, minor uncertainties in burial processes may lead to major uncertainties in steady-state concentrations, and hence, toxicity potentials.
The third aspect concerns the topic of the so-called essential elements. According to Alloway (1990) there are three criteria to determine whether or not a chemical element is biologically essential:

- the organism can neither grow nor complete its life cycle without an adequate supply of the element;
- the element cannot be wholly replaced by any other element;

– the element has a direct influence on the organism and is involved in its metabolism.

Cobalt, copper, chromium, manganese, selenium and zinc are examples of essential elements (in this case heavy metals) which are indispensable for life and may be deficient in some situations with associated problems for plants, animals and even human beings; addition of such essential elements to the environment may then have a positive effect. On the other hand, uptake of the same essential heavy metals above a certain level may have toxic impacts on plants, animals and human beings; addition of such essential elements to the environment may then have negative consequences. A discussion on how to deal with the possible positive and negative effects of essential heavy metals in the derivation of effect factors for these elements has been going on now for quite a while, but has not yet led to new derivation procedures. The derivation procedures described in this Guide for effect factors for humans, terrestrial and aquatic ecosystems give no consideration to this ongoing discussion. It is indeed to be queried if this issue is resolvable at all.

Concluding, while the method of Huijbregts (1999a) models fate and exposure routes more realistically and comprehensively than that of Hauschild & Wenzel (1998), thereby using continuous variables, there are still major uncertainties in the underlying model and its constituent parameters. Particular care should be taken if results are likely to hinge prominently on heavy metals, which score high in this method because of their persistency (this should be checked as part of the contribution analysis; see Section 5.4), This is particularly true in the case of Be and Cr. The method has, in principle, been operationalised for some 180 substances (Huijbregts, 1999a).

Hertwich (1999) has also recently calculated human toxicity potentials, employing the multimedia model CalTOX in a similar approach to Guinée *et al.* (1996) and Huijbregts (1999a; see also Huijbregts *et al*., 2000a). These so-called Toxicity Equivalence Potentials (TEPs) have been developed for air and water emissions for approximately 280 substances. The following appear to be the main differences between TEPs and the HTPs of Huijbregts (1999a; see also Huijbregts *et al*., 2000a and www.scorecard.org/env-releases/def/tep_gen.html):

– CalTOX considers more exposure routes than USES-LCA;
– the effect part of human toxicity assessment is based not on ADIs but on other human risk factors;
– CalTOX is based on American environmental data, USES-LCA on European data;
– TEPs have been developed for emissions to air and water only, HTPs for emissions to air, freshwater, seawater and ('agricultural' and 'industrial') soil;
– TEPs are available for about 280 substances, HTPs for about 180 substances.

A more thorough comparison of the human toxicity factors developed by Huijbregts (1999a; see also Huijbregts *et al.*, 2000a) and Hertwich (1999) was beyond the scope of the present study.

Hofstetter (1998) also used a multimedia model for assessing human toxicity, but now as part of an endpoint approach (*cf.* Section 4.2 for a more general description of the endpoint, or damage approach). This method, which has been developed as part of the Eco-indicator 99 approach (Goedkoop & Spriensma, 1999), has been operationalised for a limited number of substances only. Fate analysis is based on the USES 1.0 model (RIVM *et al.*, 1994), in much the same way as in Guinée *et al.* (1996). Several improvements have been introduced, however, including modeling of substance-specific dilution height. The resultant concentrations in the various environmental compartments and the ensuing, aggregated Predicted Daily Intake are then used to calculate fate factors. The damage to human health resulting from exposure to the selected substances, estimated from a range of studies, experiments and epidemiological data, is expressed in terms of Disability Adjusted Life Years (DALYs). The concept of DALYs is described in detail in Hofstetter (1998) and briefly reviewed above in the text box in Section 4.2. The fate factors and DALYs are used to calculate characterisation factors representing the damage to human health per kg substance emitted to air or water, expressed in DALYs.

3. Methods based on models and empirical relations

Jolliet & Crettaz (1997) have developed characterisation factors for human toxicity based on single-medium models and empirical measurement data to account for degradation and intermedia transport. The methods of Guinée *et al.* and Huijbregts focus more on intermedia transport modeling, that of Jolliet & Crettaz on empirical exposure relation based on measurement data or single-medium models, the main two differences being that the latter takes account of inter-substance variation in dilution volume in the final compartments air and water, and (for several substances) uses a ratio between emission flow to

air and resultant ambient concentration based on empirical data rather than modeling. All other fate routes are simulated using various partial models, however. Moreover, the methods of Guinée *et al.* (1996) and Huijbregts (1999a) apply the same model for all routes, while that of Jolliet & Crettaz (1997) employs several empirical relations derived from different unrelated sources (measurement data and single-medium models). The latter method is summarised in the following formula:

$$HTP_{i,ecomp} = \sum_{fcomp} \left[\frac{E_{i,fcomp} \times F_{i,ecomp,fcomp}}{E_{Pb,air} \times F_{Pb,air,air}} \right] \quad (4.3.7.14)$$

$$human\ toxicity = \sum_{i} \sum_{ecomp} m_{i,ecomp} \times HTP_{i,ecomp} \quad (4.3.7.15)$$

with:

HTP	Human Toxicity Potential (dimensionless);
$E_{i,fcomp}$	the effect factor, representing the human-toxic impact of substance *i* in final compartment *fcomp* and here defined as the reciprocal of the 'total acceptable world annual dose' per m^2, for air the NEC (No Effect Concentration in $kg \cdot m^{-3}$) times the total volume of air inhaled by human beings per year per m^2, and for water and soil the ADI (in kg·kg body $weight^{-1} \cdot day^{-1}$) times total body weight per m^2 and number of days per year, i.e. 365. It is expressed in different units for each *fcomp*;
$F_{i,ecom,\ fcomp}$	the fate factor for substance i, incorporating intermedia transport between emission compartment *ecomp* and final compartment *fcomp* and degradation in *fcomp* (with different units for each *fcomp*).

The method yields a single indicator result for human toxicity. For 17 substances $F_{air,air,i}$ has been calculated on the basis of empirical data, generalising the ratio between the measured concentration in Switzerland and the corresponding total emission flow to the world level. For about 100 other substances $F_{air,air,i}$ is based on the ratio between the residence time of the pollutant and the dilution height in the steady state (in $m^2 \cdot yr \cdot m^{-3}$)[1]. $F_{water,water,i}$ is based on the ratio between the residence time of the pollutant and the dilution depth in the steady state.

The ExternE project (EC, 1995a) has also assessed the potential human health impacts of air pollutants, the carcinogenic effects of trace metals, dioxins and radionuclide emissions and industrial accidents affecting members of the public. However, the format in which the dose-response functions have been published precludes derivation of characterisation factors.

Thus far, the main focus of discussion has been on the fate (and sometimes exposure/intake) element of human toxicity, leaving aside the actual effects, to which we now turn. Most methods for assessing human toxicological effects are based on the use of certain threshold values, expressed in the 'effect factor' as the reciprocal of an ADI or something similar. For such methods, the prevailing background concentration is unimportant. There is, however, a trend towards incorporating epidemiological studies on the effects actually occurring at present background concentration levels, as in the endpoint approaches of, for example, Hofstetter (1998), Goedkoop & Spriensma (1999) and EC (1995a) (*cf.* Section 4.2).

In both endpoint and midpoint approaches, there is the question of aggregation. In the case of midpoint approaches, the question is whether every ADI is equally important or whether ADIs for carcinogenic chemicals are more important than those for allergenic chemicals. Most approaches attach equal weight to every threshold value. This simple weighting procedure for human toxicity does not fulfil the ISO criteria for life cycle impact assessment (ISO, 1998a). This limitation can be overcome by breaking down human toxicity into several subcategories. Burke *et al.* (1996), for instance, has proposed dividing human-toxic substances into three categories, viz. those associated with irreversible effects, with reversible but life-threatening effects, and with reversible and non-life-threatening effects. Expert-based weighting factors of 100, 10 and 1, respectively, have been assigned to these three subcategories. Aggregation may also be based on the concept of Disability Adjusted Life Years (DALYs).

[1] In Jolliet & Crettaz this ratio is called $F_{iiecomp,fcomp}$ which is not consistent with the earlier definition.

All the methods developed to date for the impact category human toxicity suffer from a number of fundamental shortcomings. Perhaps the most serious of these is the (value) choice to effectively attach the same weight to all toxicological effects. Two other major simplifications are the assumed linearity between emissions and potential effects (*cf.* Owens, 1997b) and the complete disregard of chemical, environmental, metabolic en toxicological interactions (incl. synergy) between individual substances. Given the paucity of data currently available, the latter limitations are particularly difficult to overcome (*cf.* Huijbregts *et al.*, 2000a).

PROSPECTS

In the future, further progress is anticipated on Impact assessment modeling to the various endpoints, viz. damage to human health, damage to ecosystem health, etc. (*cf.* Figure 4.2.2); see Section 4.2 for a more extensive discussion of this subject).

CONCLUSIONS

Despite these fundamental limitations, which are not specific to LCA but are also encountered in RA, we still consider it possible and useful to recommend a baseline method for the impact category human toxicity. It is possible, since there are practical methods available reflecting, as well as is feasible, the state-of-the-art in human toxicity assessment; and useful, since the alternative would be that potentially human-toxic emissions would otherwise be only qualitatively flagged, clouding final assessment, or even omitted altogether.

As the baseline characterisation method for human toxicity we thus recommend using the method of Huijbregts (1999a), based on fate modeling with USES-LCA. USES-LCA treats intermedia transport most realistically and comprehensively, using continuous variables and a multimedia model. In our view this is superior to the 'rule of thumb' method developed by Hauschild & Wenzel (1998) and the mix of models and measured data used by Jolliet & Crettaz (1997) (criterion 8 for the selection of baseline characterisation methods). Those of Huijbregts (1999a) and Hertwich (1999) appear to be very similar. Both methods are operational, while the method of Hertwich (1999) includes more substances (criterion 11). However, Huijbregts' method can also be used for emissions to soil. This is especially important for LCAs on agricultural products, where emissions to soil are significant. Huijbregts' method encompasses both human toxicological and ecotoxicological effects (see Section 4.3.8), moreover, enabling both impact categories to be assessed using the same method.

The method of Huijbregts is thus preferable to that of Hertwich (1999) and in this Guide is therefore recommended as the baseline. An infinite time horizon is thereby taken (criterion 7) and a global spatial scale (criterion 10). It should be borne in mind, however, that the list of 180 substances provided by Huijbregts (1999a) represents merely a very small subset of all known and unknown toxic substances. If the case study involves substances suspected of contributing to human toxicity for which HTPs are not provided in this Guide, such factors should be calculated. If this is not feasible, an attempt should be made to estimate HTPs for these substances based on similar or related substances for which HTPs are available (by extrapolation, for example; *cf.* Section 4.3.17). It is not recommended to use the 'old' characterisation factors that ignore fate, such as those developed by Heijungs *et al.* (1992), even if these are available for more substances than the new, fate-based factors. Fate is a particularly important consideration in the context of the human toxicity of chemical pollutants and its exclusion might lead to misleading results.

The baseline characterisation method for human toxicity, using the characterisation factor $HTPs_{\infty, global}$, is evaluated with respect to the (ISO-based) criteria in Table 4.3.7.1.

Table 4.3.7.1: Evaluation of the baseline characterisation method for human toxicity, using the characterisation factor $HTP_{\infty,global}$, with respect to the (ISO-based) criteria of Table 4.3.1.

	criterion	evaluation
1.	scientifically and technically valid	partly: although fate and exposure/intake calculations are based on well understood environmental mechanisms, actual toxicological effects are assessed via a very crude aggregation of very different effects (from skin irritation to mortality)
2.	environmentally relevant	yes, the category indicator represents risks to humans (near-endpoint level)

	criterion	evaluation
3.	internationally accepted	no; however, USES-LCA is very similar to the model EUSES, supported by the European Union (EUSES is in fact based on USES)
4.	value-choices and assumptions	yes; the choice of valuing all effects equally is particularly debatable
5.	focal point in environmental mechanism	(nearly) endpoint
6.	linearity	yes
7.	time span	eternity
8.	fate, exposure/intake and effects	fate, exposure/intake and effects included
9.	less is better	yes; below-threshold effects are also included
10.	time- and location-independent	yes, although in fact the HTPs are European factors
11.	operational	yes, for 180 substances emitted to air, water and soil
12.	uncertainty margins	presumably about the same as for the other category indicators that include fate, discussed above

The following methods are included in this Guide as options for sensitivity analysis, particularly for LCA studies in which results for this impact category are dominated by metals and volatile, persistent halogenated organics:

- The HTPs for the time horizons 20, 100 and 500 years at the global scale: Huijbregts *et al.* (2000).
- The HTPs for an infinite time horizon at the continental scale (i.e. excluding global effects): Huijbregts *et al.* (2000).

Recommendations for extended LCAs:

- If the case study involves substances suspected of contributing to human toxicity for which HTPs are not provided in this Guide, such factors should be calculated. If this is not feasible, an attempt should be made to estimate HTPs for these substances based on similar or related substances for which HTPs are available (by extrapolation, for example; *cf.* Section 4.3.17).

method status	characterisation method/factor	reference
baseline	$HTP_{\infty,global}$	Huijbregts, 1999a
alternative 1	$HTP_{20,global}$	Huijbregts *et al*., 2000
alternative 2	$HTP_{100,global}$	Huijbregts *et al*., 2000
alternative 3	$HTP_{500,global}$	Huijbregts *et al*., 2000
alternative 4	$HTP_{\infty,continental}$	Huijbregts *et al*., 2000
additional	–	–
variant	–	Hauschild & Wenzel, 1998; Guinée *et al*., 1996; Hertwich, 1999; Jolliet & Crettaz, 1997

RESEARCH RECOMMENDATIONS

Short-term research:

- It is recommended to undertake a detailed comparison of the factors of Huijbregts (1999a) and Hertwich (1999) to establish which differences are due to differences in data and which to the use of different fate and effect models.
- It is recommended to develop characterisation factors for group parameters according to the procedures described in Section 3.6.
- It is recommended to establish a 'helpdesk' to provide support for calculation of characterisation factors for substances for which these factors are currently lacking.

Long-term research:

- Further research to explore the potential for using Disability Adjusted Life Years (DALYs) in LCA is recommended, especially for this impact category. The possible integration of toxic effects on human health with other human health effects such as the effects of photo-oxidants, radiation,

casualties, etc., should also be investigated. The DALY concept might be very suitable for the purpose of such integration.

- The EDIP approach, employing a database of impacts in the working environment per kilogram of goods produced for specific economic activities may represent a valuable addition to LCA (Schmidt & Brunn Rasmussen, 1999). This database comprises mainly Danish data, however, and research should be undertaken to establish a similar database with European data.

4.3.8 Ecotoxicity

TOPIC

This impact category covers the impacts of toxic substances on aquatic, terrestrial and sediment ecosystems. The area of protection is the natural environment (and natural resources) (see Figure 4.2.2).

For this impact category a variety of characterisation methods are available, providing characterisation factors generally that are referred to as 'ecotoxicity potentials' (ETPs). As in the previous section, on human toxicity, the following general formula will here be used as a point of departure for describing and discussing these different methods:

$$ETP_{i,ecomp} = \sum_{fcomp} F_{i,ecomp,fcomp} \times E_{i,fcomp} \qquad (4.3.8.1)$$

with:

$ETP_{i,ecomp}$ the ecotoxicity potential: the contribution to ecotoxicity of a unit emission of substance *i*, to emission compartment *ecomp*. Most methods distinguish several subcategories, such as AETP for aquatic ecotoxicity, TETP for terrestrial ecotoxicity, etc.;

$F_{i,ecomp,fcomp}$ a 'fate factor', representing intermedia transport of substance *i* from emission compartment *ecomp* to final (sub)compartment *fcomp*, and degradation within compartment *ecomp*; in some methods intermedia transport is indicated separately by $f_{i,ecomp,fcomp}$ and (bio)degradation by BIO_i;

$E_{i,fcomp}$ an 'effect factor', representing the toxic effect of exposure of a given ecosystem to substance *i* in compartment *fcomp*.

The ETP is often defined relative to a reference substance. As a formula:

$$ETP_{i,ecomp} = \frac{\sum_{fcomp} F_{i,ecomp,fcomp} \times E_{i,fcomp}}{\sum_{fcomp} F_{refi,refecomp,fcomp} \times E_{refi,fcomp}} \qquad (4.3.8.2)$$

with the symbols similar to the above. The choice of the reference substance is arbitrary.

DEVELOPMENTS IN THE LAST DECADE

Ecotoxicological impact, together with human toxicological impact, is the impact category for which fate and particularly intermedia transport are most important. Toxic substances do not generally remain in the environmental compartment into which they are emitted, but tend to spread to other compartments, where they may do more damage. Airborne insecticides will also settle out in waterways, for example, where they may cause (severe) harm to aquatic organisms. As with human-toxic impacts, many authors therefore stress that it is essential to properly incorporate intermedia transport in LCA models of ecotoxicological impact (e.g. Heijungs *et al.*, 1992; Lindfors *et al.*, 1995c; Jolliet, 1996; Guinée *et al.*, 1996; Jolliet & Crettaz, 1997; Hauschild & Wenzel, 1998).

Heijungs *et al.* (1992)

In Heijungs *et al.* (1992) this impact category covered only toxic emissions to the environmental compartments water and soil. Emissions to water were considered to be toxic to aquatic ecosystems only and emissions to soil toxic to terrestrial ecosystems only. Separate characterisation factors were thus defined for emissions to water and soil:

$$ECA = \frac{1}{MTC_a} \qquad (4.3.8.3)$$

$$ECT = \frac{1}{MTC_t} \qquad (4.3.8.4)$$

where ECA and ECT are, respectively, the characterisation factor for aquatic and for terrestrial ecosystems (m^3 water·mg^{-1} substance and kg soil·mg^{-1} substance) and MTC_a and MTC_t are the Maximum Tolerable Concentrations for water and soil, derived according to a method developed by EPA and modified by RIVM (EPA, 1984; Van de Meent *et al.*, 1990; Slooff, 1992). These MTCs represent the concentration considered to protect 95% of the species in an ecosystem. They are based on ecotoxicological data on the species sensitivity to chemical substances, using the modified EPA method, which employs safety factors based on the number of test species included (see, *inter alia*, RIVM *et al.*, 1994).

The characterisation factors used to assess and aggregate the interventions for the impact categories aquatic and terrestrial ecotoxicity are:

$$Aquatic\ ecotoxicity = \sum_i \left(ECA_i \times m_{w,i}\right) \qquad (4.3.8.5)$$

$$Terrestrial\ ecotoxicity = \sum_i \left(ECT_i \times m_{s,i}\right) \qquad (4.3.8.6)$$

where $m_{w,i}$ (mg) is the amount of substance *i* emitted to water and $m_{s,i}$ the amount of substance *i* emitted to soil. The indicator results *terrestrial ecotoxicity* and *aquatic ecotoxicity* are expressed in kg soil and m^3 water. They can be interpreted as the quantity of terrestrial or aquatic ecosystem polluted to the MTC. As with the impact category "human toxicity", Heijungs *et al.* again stress that these characterisation factors for ecotoxicity should be considered as no more than indicative until such time as a better method is developed.

In line with these recommendations, we here consider only methods that include degradation and intermedia transport, which supersede the provisional method developed by Heijungs *et al.* (1992)[1]. (See text box.) Several such methods are described in the literature, falling roughly into three groups:

The architects of the intermedia transport-based characterisation factors for human toxicity described in the previous section have developed such factors for ecotoxicity, with methods once more falling roughly into three groups:

1. methods in which degradation and intermedia transport is based on simple rules of thumb (e.g. Hauschild & Wenzel, 1998)
2. methods in which degradation and intermedia transport is based on models (e.g. Guinée *et al.*, 1996; Huijbregts, 1999a)
3. methods in which degradation and intermedia transport is based on models and empirical relations (e.g. Jolliet & Crettaz (1997).

[1] Heijungs *et al.* stressed in 1992, already, that fate should be included; at the time, however, this could be done only provisionally.

In this section we consider only those aspects of these methods and models of specific relevance for the impact category 'ecotoxicity', referring the reader for background to Section 4.3.7 on 'human toxicicity', as necessary. As in the previous section, these methods are discussed below in terms of a common notation. For instance, F always denotes a fate factor, although a variety of symbols and terms are encountered in the literature. Moreover, the precise operationalisation and , consequently, the units used for broadly similar variables may differ from method to method.

1. Methods based on rules of thumb

Based on rule of thumb, Hauschild & Wenzel (1998) have developed characterisation factors[1] incorporating intermedia transport for three[2] different impact categories:

$$AETP_{i,ecomp,acute} = f_{i,ecomp,fcomp} \times E_{i,aqua/acute} \quad (4.3.8.7)$$

$$AETP_{i,ecomp,chronic} = f_{i,ecomp,fcomp} \times BIO_i \times E_{i,aqua/chronic} \quad (4.3.8.8)$$

$$TETP_{i,ecomp,chronic} = f_{i,ecomp,fcomp} \times BIO_i \times E_{i,terr} \quad (4.3.8.9)$$

with:

$AETP_{i,ecomp,acute/chronic}$ the Aquatic EcoToxicity Potential of substance *i* emitted to emission compartment *ecomp*, with an indication of the type of impact considered (*acute* or *chronic*);

$TETP_{i,ecomp,chronic}$ the Terrestrial EcoToxicity Potential of substance *i* emitted to emission compartment *ecomp*, with an indication of the type of impact considered (in this case only *chronic*);

$f_{i,ecomp,fcomp}$ the intermedia transport factor, the fraction of substance *i* emitted to emission compartment *ecomp* that reaches final compartment *fcomp* as a result of environmental transport. The factor is based on a simple rules of thumb tather than on a full fate model; moreover, it is not a continuous variable but assumes only a limited number of values such as 0.2 and 1;

BIO_i the biodegradability factor, to be chosen from one of three classes depending on the substance *i* involved;

$E_{i,fcomp}$ the effect factor, representing the (acute and/or chronic) toxic impact of substance *i* on aquatic or terrestrial ecosystems (respectively) in final compartment *fcomp*; depending on *fcomp* and the type of effect, it is the reciprocal of a PNEC[3] (Predicted No Effect Concentration).

Three different emission compartments are considered by Hauschild & Wenzel: air, water and soil. The effect compartments considered are water and soil. In the aquatic compartment acute and chronic effects are considered separately. Together this results in seven different ecotoxicity potentials per substance.

Hauschild & Wenzel's method yields three separate indicator results for ecotoxicity:

one for chronic effects in the aquatic compartment:

$$aquatic\ chronic\ ecotoxicity = \sum_i \sum_{ecomp} m_{i,ecomp} \times AETP_{i,ecomp,chronic} \quad (4.3.8.10)$$

one for acute effects in the aquatic compartment:

$$aquatic\ acute\ ecotoxicity = \sum_i \sum_{ecomp} m_{i,ecomp} \times AETP_{i,ecomp,acute} \quad (4.3.8.11)$$

and one for chronic effects in soil compartment:

$$terrestrial\ chronic\ ecotoxicity_i = \sum_i \sum_{ecomp} m_{i,ecomp} \times TETP_{i,ecomp,chronic} \quad (4.3.8.12)$$

[1] Again, for reasons of terminological consistency we here employ a number of terms as originally defined by Hauschild & Wenzel.

[2] Acute toxicity to (the micro-organisms in) wastewater treatment plants has been suggested as a fourth impact category, but waste water treatment plants are considered to be part of the economic system in this Guide (*cf.* Section 3.2), we have restricted the discussion to the three environmental impact categories acute aquatic ecotoxicity, chronic aquatic ecotoxicity and chronic terrestrial ecotoxicity.

[3] The PNEC is based on species-specific ecotoxicological data derived by a variety of methods, in some cases policy targets (see Hauschild & Wenzel, 1998).

AETP and TETP are defined as specified above, and $m_{i,ecomp}$ is the emission of substance *i* to compartment *ecomp*.
These indicator results are not aggregated to a single indicator result.

Overall, it can be concluded that the method of Hauschild and Wenzel (1998) models degradation and intermedia transport only very rudimentarily.

2. Model-based methods

Guinée *et al.* (1996) have also developed characterisation factors for aquatic and terrestrial ecotoxicity impacts that take some account of degradation and intermedia transport. The method they employed is basically the same as that used for human toxicity impacts and is based once more on the USES 1.0 model, which includes Simplebox 1.0 as a model. A Predicted Environmental Concentration (PEC)[1] is calculated for the compartments water, agricultural soil and industrial soil resulting resulting from a given constant emission to air, water, agricultural or industrial soil (i.e. soil used for agricultural and industrial purposes). The PEC in water is divided by the Predicted No Effect Concentration[2] (PNEC) for aquatic ecosystems, the PEC in soil by the PNEC for terrestrial ecosystems. Finally, the score is divided by the score for the emission of a reference substance, resulting in:

$$AETP_{i,ecompi} = \frac{PECwater_{ecomp,i} \times Eaqua_i}{PECwater_{water\ 1,4-dichlorobenzene} \times Eaqua_{1,4-dichlorobenzene}} \quad (4.3.8.13)$$

$$TETP_{i,ecomp} = \frac{PECagricultural\ soil_{i,ecomp} \times Eterr_i}{PECagricultural\ soil_{industrial\ soil,1,4-dichlorobenzene} \times Eterr_{1,4-dichlorobenzene}} \quad (4.3.8.14)$$

with:

$AETP_{i,ecomp}$ the Aquatic EcoToxicity Potential due to emission of 1000 kg of substance *i* per day (flux) to initial emission compartment *ecomp* (dimensionless);

$TETP_{i,ecomp}$ the Terrestrial EcoToxicity Potential, defined in a parallel manner (dimensionless);

$PECwater_{i,ecomp}$ the predicted concentration of substance *i* in water due to the emission to compartment *ecomp*; $PECwater_{1,4\text{-diochlorobenzene, ecomp}}$ is the same predicted concentration but for 1,4-dichlorobenzene ($kg.m^{-3}$).

$PECagricultural\ soil_{i,ecomp}$ the predicted concentration of substance *i* in agricultural soil due the emission to compartment *ecomp*; $PECagricultural\ soil_{1,4\text{-diochlorobenzene, ecomp}}$ is the same predicted concentration but for 1,4-dichlorobenzene (dimensionless).

$Eaqua_i$ the effect factor representing the toxic impact of substance *i* on aquatic ecosystems, here the reciprocal of the PNEC for terrestrial ecosystems; $Eaqua_{1,4\text{-diochlorobenzene}}$ is the same effect factor but for 1,4-dichlorobenzene ($m^3.kg^{-1}$).

$Eterr_i$ the effect factor representing the toxic impact of substance *i* on terrestrial ecosystems, here the reciprocal of the PNEC for terrestrial ecosystems; $Eterr_{1,4\text{-diochlorobenzene}}$ is the same effect factor but for 1,4-dichlorobenzene (dimensionless).

The compartments considered are air, water and agricultural and industrial soil, resulting in four AETPs and four TETPs per substance. The result is one indicator result for aquatic ecotoxicity and one for terrestrial ecotoxicity:

$$aquatic\ ecotoxicity = \sum_i \sum_{ecomp} mi_{i,ecomp} \times AETP_{i,ecomp} \quad (4.3.8.15)$$

$$terrestrial\ ecotoxicity = \sum_i \sum_{ecomp} m_{i,ecomp} \times TETP_{i,ecomp} \quad (4.3.8.16)$$

[1] $PEC_{i,ecomp}$ is an alternative formulation of $F_{i,ecomp,fcomp}$.
[2] In this case the PNEC was defined similarly to the MTC: the concentration held to protect 95% of the species in an ecosystem. This PNEC was also based on species-specific ecotoxicological data using a variety of methods; see VROM (1997).

where $AETP_{i,ecomp}$ and $TETP_{i,ecomp}$ are as defined above and $m_{i,ecomp}$ is the amount of substance *i* emitted to compartment *ecomp* (kg).

Huijbregts (1999a, 2000) developed the USES-LCA model to calculate not only new characterisation factors for human toxicity but also for five subcategories of ecotoxicity: freshwater aquatic, marine aquatic, freshwater sediment, marine sediment and terrestrial, each of them for different time horizons. The main differences from Guinée *et al.* have already been described (Section 4.3.7). The factors proposed by Huijbregts are:

$$FAETP_{i,ecomp} = \frac{PEC_{i,ecomp,freshwater} \times E_{i,freshwater}}{PEC_{1,4-dichlorobenzene,freshwater,freshwater} \times E_{1,4-dichlorobenzene,freshwater}} \qquad (4.3.8.17)$$

with:

$FAETP_{i,ecomp}$ the Freshwater Aquatic EcoToxicity Potential of substance *i* emitted to emission compartment *ecomp*;

This is extended to the other impact categories, giving the following scheme:

impact (sub)category	characterisation factor
freshwater ecotoxicity	$FAETP_{i,ecomp}$
freshwater sediment ecotoxicity	$FSETP_{i,ecomp}$
marine aquatic ecotoxicity	$MAETP_{i,ecomp}$
marine sediment ecotoxicity	$MSETP_{i,ecomp}$
terrestrial ecotoxicity	$TETP_{i,ecomp}$

These ecotoxicity potentials are based on a PEC/PNEC ratio, weighted as necessary on the basis of the volume (water) or weight (soil and sediment) of the compartments/scales concerned (see also Section 4.3.7 and, for a more detailed explanation, Huijbregts, 1999a).

For these ecotoxicity potentials, Huijbregts *et al.* (2000) again used scenario analysis to assess the sensitivity of the model to different time and spatial horizons. For a discussion of this topic we refer the reader to Section 4.3.7, for a detailed description of calculation procedures and models used to Huijbregts (2000).

As in the case of human toxicity, the method of Huijbregts (1999a) takes degradation and intermedia transport routes into account more realistically than that of Hauschild & Wenzel (1998). The method is operational for 181 substances, in principle.

3. Methods based on models and empirical relations

Jolliet & Crettaz (1997) also developed their 'critical surface time' method for ecotoxicity, calculating characterisation factors for aquatic and terrestrial ecotoxicity:

$$AETP_{i,ecomp} = E_{i,aqua} \times F_{i,ecomp,w} \qquad (4.3.8.18)$$

$$TETP_{i,ecomp} = E_{i,terr} \times F_{i,ecomp,s} \qquad (4.3.8.19)$$

$$aquatic\ ecotoxicity = \sum_i \sum_{ecomp} m_{i,ecomp} \times AETP_{i,ecomp} \qquad (4.3.8.20)$$

$$terrestrial\ ecotoxicity = \sum_i \sum_{ecomp} m_{i,ecomp} \times TETP_{i,ecomp} \qquad (4.3.8.21)$$

with:

$E_{i,aqua}$ the effect factor representing the toxic impact of substance *i* on aquatic ecosystems and here defined as the reciprocal of the PNEC[1] (Predicted No Effect Concentration for aquatic ecosystems);

[1]The PNECs of Jolliet & Crettaz (1997) are based on species-specific ecotoxicological data, similarly to the MTCs of Heijungs *et al.* 1992; in this case the modified EPA method was used (see, *inter alia,* RIVM *et al.*, 1994).

$E_{i,terr}$ the effect factor representing the toxic impact of substance *i* on terrestrial ecosystems and here defined as the reciprocal of the PNEC[1] (Predicted No Effect Concentration for terrestrial ecosystems);

$F_{i,ecomp,water}$ the fate factor for water, the fraction of substance *i* emitted to emission compartment *ecomp* that reaches the final compartment *water*;

$F_{i,ecomp,soil}$ the fate factor for soil, the fraction of substance *i* emitted to emission compartment *ecomp* that reaches the final compartment *soil*;

$m_{i,ecomp}$, $AETP_{i,ecomp}$ and $TETP_{i,ecomp}$ are as above.

These fate facors are based on the same intermedia transfer factors as for human toxicity.

Thus far, the main focus of discussion has been on the fate element of ecotoxicity. We now turn to the actual effects. Most methods for assessing ecotoxicological effects are based on the use of certain threshold values, expressed in the 'effect factor; as the reciprocal of a PNEC os something similar. For such methods, the prevailing background concentration is unimportant. There is, however, a trend towards incorporating concentration-effect curves, or the slopes of such curves, in which the effect factor is dependent upon the background concentration.
This is the case, for example, in the endpoint approach used in the Eco-indicator 99 (Goedkoop & Spriensma, 1999; *cf.* Section 4.2). This is based on the notion of PAFs (Potentially Affected Fraction), the fraction of species that, given an environmental concentration, is exposed above the No Observed Effect Concentration (NOEC) (Van de Meent, 1999). The higher the concentration, the greater the percentage of species considered to be affected. PAFs are based on substance-specific species-sensitivity distributions, in turn are based on the NOECs for these substances for different species. The PAF-concentration curve typically has a log-log shape (see Figure 4.3.8.1).

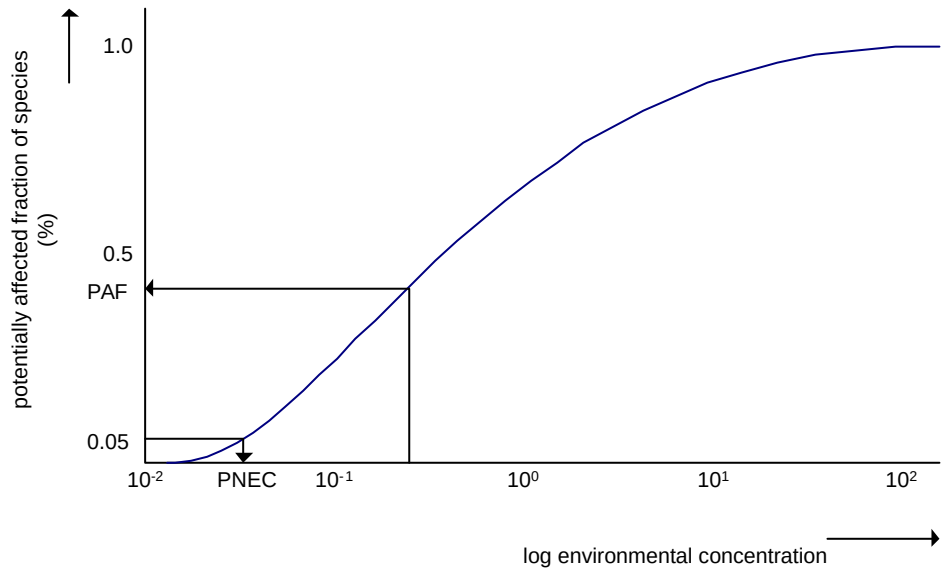


Figure 4.3.8.1: The PAF curve (source: Van de Meent, 1999).

The PAF and the PNEC are both based on the same curve (see Figure 4.3.8.1). To apply the PAF concept requires additional data, however, for example on current environmental concentrations. As these data are still lacking for most substances, the concept still has little practical use. The fact that the PAF provides a measure of toxicity that is comparable between substances is considered to be its major advantage. However, specific rules of addition hold for combining PAFs, which are difficult to transpose to LCA, because as a baseline LCA integrates over space and time and these addition rules for combined toxicity are of course only valid when the substances are present at the same place at the same time.

While ecotoxicological impact assessment is based on PAFs in the Eco-indicator 99 method, fate and exposure modeling is based on EUSES, a European expansion and update of USES 1.0. The method is operational for 46 substances emitted to air, water and soil.

The methods used to derive characterisation factors for ecotoxicological impacts very much resemble those employed for human toxicological impacts. There is one major difference, however. In the case of human toxicity the ADI is often taken as a reference level for risk evaluation. The ADI (Acceptable Daily Intake) represents an acceptable risk level for one species: humans, based on extrapolation of a range of known human toxicological effects. In the case of ecotoxicity a PNEC (Predicted No Effect Concentration) or MTC (Maximum Tolerable Concentration) is used as a reference level to evaluate the risk. These levels represent an acceptable risk level for ecosystems and are based on extrapolation of selected toxic effects on a few selected species to the overall 'toxic impact' on an entire ecosystem.

PROSPECTS

In the future, further progress is anticipated on Impact assessment modeling to the various endpoints, viz. damage to human health, damage to ecosystem health, etc. (*cf.* Figure 4.2.2); see Section 4.2 for a more extensive discussion of this subject). In this respect the PAF method seems promising, but requires further work and operationalisation.

At the moment the EPA is working on a list of so-called 'PBT' chemicals that are persistent, bioaccumulative and toxic, to be used to focus source-reduction and recycling measures. To compile the list some 4000 chemicals are being screened and the resultant PBT information stored in a database. This database might perhaps in the future be used in combination with the data of Huijbregts (1999a) and the USES-LCA data to gain an approximate indication of human and ecotoxicological characterisation factors for a large group of substances.

CONCLUSIONS

Given the foregoing considerations, for the same reasons as for impact category human toxicity we recommend using Huijbregts (1999a), based on fate modeling with USES-LCA, as the baseline characterisation method for ecotoxicity. USES-LCA treats intermedia transport in most realistically and comprehensively, using continuous variables. The fact that the method of Huijbregts also comprises factors for human toxicity (see Section 4.3.7) is an additional advantage, allowing the two impact categories to be assessed with a similar fate model. An infinite time horizon is thereby adopted (criterion 7), and the global scale as spatial horizon (criterion 10). As with human toxicity, however, it should be borne in mind that the list of 180 substances provided by Huijbregts (1999a) represents merely a very small subset of all known and unknown toxic substances. If the case study involves substances suspected of contributing to ecotoxicity for which no characterisation factors are provided in this Guide, such factors should be calculated. If this is not feasible, an attempt should be made to estimate such factors based on similar or related substances for which they are available (by extrapolation, for example; *cf.* Section 4.3.17). It is not advised to use 'old' characterisation factors that ignore fate, such as those developed by Heijungs *et al.* (1992), even if these are available for more substances than the new, fate-based factors. Fate is a particularly important consideration in the context of the ecotoxicity of chemical pollutants and its exclusion might lead to misleading results.
The baseline characterisation method for ecotoxicity, using the characterisation factor $ETP_{\infty,global}$, is evaluated with respect to the (ISO-based) criteria in Table 4.3.8.1.

Table 4.3.8.1: Evaluation of the baseline characterisation method for ecotoxicity, using the characterisation factor $ETP_{\infty,global}$, with respect to the (ISO-based) criteria of Table 4.3.1.

	criterion	evaluation
1	scientifically and technically valid	partly: although fate calculations are based on well understood environmental mechanisms, actual toxicological effects are assessed via a very crude aggregation of very different effects (from reduced fertility to mortality) among species and ecosystems
2	environmentally relevant	yes, the category indicator represents risks to ecosystems (near-endpoint level)
3	internationally accepted	no; however, USES-LCA is very similar to the model EUSES, supported by the European Union (EUSES is in fact based on USES)
4	value-choices and assumptions	yes; the choice of valuing all effects and species equally is particularly debatable

	criterion	evaluation
5.	focal point in environrmental mechanism	(nearly) endpoint
6.	linear	yes
7.	time span	eternity
8.	fate, exposure/intake and effects	fate included, exposure/intake not relevant, effects included
9.	less is better	yes; below-threshold effects are also included
10.	time- and location-independent	yes, although in fact the ETPs are European factors
11.	operational	yes, for 180 substances emitted to air, water and soil
12.	uncertainty margins	presumably about the same as for the other category indicators that include fate (method group 2)

The following methods are included in this guide as options for sensitivity analysis, particularly for LCA studies in which results for this impact category are dominated by metals and volatile, persistent halogenated organics:

- The FAETPs, MAETPs, FSETPs, MSETPs and TETPs for the time horizons 20, 100 and 500 years at the global scale: Huijbregts *et al.* (2000).
- The FAETPs, MAETPs, FSETPs, MSETPs and TETPs for an infinite time horizon at the continental scale (i.e. excluding global effects): Huijbregts *et al.* (2000).

Recommendations for extended LCAs:

- If the case study involves substances suspected of contributing to ecotoxicity for which FAETPs, MAETPs, FSETPs, MSETPs and TETPs are not provided in this Guide, such factors should be calculated. If this is not feasible, an attempt should be made to estimate ETPs for these substances based on similar or related substances for which these ETPs are available (by extrapolation, for example; *cf.* Section 4.3.17).

Freshwater aquatic ecotoxicity:

method status	characterisation method/factor	reference
baseline	$FAETP_{\infty,global}$	Huijbregts, 1999a
alternative 1	$FAETP_{20,global}$	Huijbregts *et al*., 2000
alternative 2	$FAETP_{100,global}$	Huijbregts *et al*., 2000
alternative 3	$FAETP_{500,global}$	Huijbregts *et al*., 2000
alternative 4	$FAETP_{\infty,continental}$	Huijbregts *et al*., 2000
additional	–	–
variant	–	Hauschild & Wenzel, 1998; Guinée *et al*., 1996; Jolliet & Crettaz, 1997

Marine ecotoxicity:

method status	characterisation method/factor	reference
baseline	$MAETP_{\infty,global}$	Huijbregts, 1999a
alternative 1	$MAETP_{20,global}$	Huijbregts *et al*., 2000
alternative 2	$MAETP_{100,global}$	Huijbregts *et al*., 2000
alternative 3	$MAETP_{500,global}$	Huijbregts *et al*., 2000
alternative 4	$MAETP_{\infty,continental}$	Huijbregts *et al*., 2000
additional	–	–
variant	–	–

Terrestrial ecotoxicity:

method status	characterisation method/factor	reference
baseline	$TETP_{\infty,global}$	Huijbregts, 1999a
alternative	$TETP_{20,global}$	Huijbregts *et al*., 2000
	$TETP_{100,global}$	Huijbregts *et al*., 2000
	$TETP_{500,global}$	Huijbregts *et al*., 2000
	$TETP_{\infty,continental}$	Huijbregts *et al*., 2000
additional	–	–
variant	–	Hauschild & Wenzel, 1998; Guinée *et al*., 1996; Jolliet & Crettaz, 1997

Freshwater sediment ecotoxicity:

method status	characterisation method/factor	reference
baseline	$FSETP_{\infty,global}$	Huijbregts, 1999a
alternative 1	$FSETP_{20,global}$	Huijbregts *et al*., 2000
alternative 2	$FSETP_{100,global}$	Huijbregts *et al*., 2000
alternative 3	$FSETP_{500,global}$	Huijbregts *et al*., 2000
alternative 4	$FSETP_{\infty,continental}$	Huijbregts *et al*., 2000
additional	–	–
variant	–	–

Marine sediment ecotoxicity:

method status	characterisation method/factor	reference
baseline	$MSETP_{\infty,global}$	Huijbregts, 1999a
alternative 1	$MSETP_{20,global}$	Huijbregts *et al.*, 2000
alternative 2	$MSETP_{100,global}$	Huijbregts *et al.*, 2000
alternative 3	$MSETP_{500,global}$	Huijbregts *et al.*, 2000
alternative 4	$MSETP_{\infty,continental}$	Huijbregts *et al*., 2000
additional	–	–
variant	–	–

RESEARCH RECOMMENDATIONS

Long-term research:

- The potential for integrating the method of Huijbregts (1999a) with the so-called PAF approach should be investigated.
- The potential for using the PAF approach to aggregate ecotoxicological with such other effects as eutrophication and acidification should be further investigated.
- This database might perhaps in the future be used in combination with the data of Huijbregts (1999a) and the USES-LCA data to calculate approximate human and ecotoxicological characterisation factors for a larger group of substances.

4.3.9 Photo-oxidant formation

TOPIC

Photo-oxidant formation is the formation of reactive chemical compounds such as ozone by the action of sunlight on certain primary air pollutants. These reactive compounds may be injurious to human health and ecosystems and may also damage crops. The relevant areas of protection are human health, the man-made environment, the natural environment and natural resources (Udo de Haes *et al.,* 1999).

Photo-oxidants may be formed in the troposphere under the influence of ultraviolet light, through photochemical oxidation of Volatile Organic Compounds (VOCs) and carbon monoxide (CO) in the presence of nitrogen oxides (NO_x). Ozone is considered the most important of these oxidising compounds, along with peroxyacetylnitrate (PAN). Photo-oxidant formation, also known as summer smog, Los Angeles smog or secondary air pollution, contrasts with winter smog, or London smog, which is characterised by high levels of inorganic compounds, mainly particles, carbon monoxide and sulphur

compounds. This latter type of smog causes bronchial irritation, coughing, etc. Winter smog, as far as considered in this Guide, is part of human toxicity.

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)
The numerous atmospheric species of VOC vary widely in their contribution to photo-oxidant formation. In Heijungs *et al.* (1992) Photochemical Ozone Creation Potentials (POCPs) were used as a characterisation factor to assess and aggregate the interventions for the impact category photo-oxidant formation:

$$Photo-oxidant\ formation = \sum_i POCP_i \times m_i \qquad (4.3.9.1)$$

where m_i (kg) is the mass of substance *i* released, $POCP_i$ the photochemical ozone creation potential of the substance and *Photo-oxidant formation* is the indicator result, which is expressed in kg ethylene-equivalents.
In Heijungs *et al.* (1992) it was proposed to flag NO_x emissions to remind practitioners of their crucial relevance to photo-oxidant formation.
In most current LCA studies the available data set will not include specific VOC species, but only groups of hydrocarbons and methane. In such cases Heijungs *et al.* (1992) recommended deriving average POCPs for these groups from the arithmetical average of the data for individual VOCs.
The POCP is calculated as the estimated quantity of ozone formed photochemically by a given VOC, using a model to calculate ozone formation in the presence and absence of this compound (UNECE, 1990). This means that, unlike ODPs and GWPs, the POCPs calculated by Derwent & Jenkin (1990) are based on an 'average' rather than a 'marginal' approach. Heijungs *et al.* (1992) suggested that it would be more attractive to calculate POCPs on the basis of a marginal change in emissions, as with GWPs and ODPs. One of the advantages of a marginal approach is that NO_x would also be included, as explained below, in contrast to Derwent & Jenkin's method, which covers only VOCs. Although NO_x acts as a catalyst in the chemical reactions involved in photochemical smog formation and is therefore not consumed, its background concentration affects the course of ozone production. As NO_x is also removed from the atmosphere in the form of (acidic, eutrophying) precipitation and deposition, photochemical reactions will only be maintained if there is a continuous supply thereof. In the methods currently used to estimate POCPs the emission-effect curve is assumed to be a straight line through the origin. This is an extremely coarse approximation of the actual situation and may introduce errors for certain substances, including NO_x. In principle, it would be possible to calculate a POCP for NO_x. However, setting NO_x emissions to zero for the reference part of the calculation outlined above would lead to absurd results, as there would be no ozone production at all without NO_x. The emission-effect curve of NO_x is far from linear. If POCPs were calculated on the basis of marginal changes in emissions, a POCP could be derived for NO_x.

Today, three methods are available for comparing the ozone creation potential of different species of VOC, based on:
1. POCPs (e.g. Derwent & Jenkin, 1990, Derwent *et al.*, 1998)
2. fate factors (Hofstetter, 1998)
3. Incremental Reactivity (e.g. Carter, 1994)

Methods based on POCPs
Photochemical Ozone Creation Potentials (POCPs) were originally developed to assess various emission scenarios for volatile organic compounds (Derwent & Jenkin, 1990). A UN protocol defined the POCP of a VOC as the ratio between the change in ozone concentration due to a change in the emission of that VOC and the change in the ozone concentration due to a change in the emission of ethylene (UNECE, 1990). Expressed as a formula:

$$POCP_i = \frac{a_i / b_i}{a_{C_2H_4} / b_{C_2H_4}} \qquad (4.3.9.2)$$

where a_i is the change in ozone concentration due to a change in the emission of VOC *i* and b_i the integrated emission of VOC *i* up to that time, with the denominator containing these parameters for ethylene, the reference substance.

The POCPs of Derwent & Jenkin (1990), used as characterisation factors in Heijungs *et al.* (1992), were updated in 1996, 1998 and 1999 (Derwent *et al.*, 1996, 1998; Jenkin & Hayman, 1999). These figures are based on a 5-day trajectory model of VOC transportation above Europe. In the first update POCPs for NO, NO_2 and several other inorganic substances were also included, in contrast to the Derwent & Jenkin POCPs of 1990. However, it is not clearly explained how these were calculated. In the last two updates only the POCPs for VOCs were revised, using a new calculation method. The original POCPs for these compounds were based on the difference in ozone formation with and without the VOC in question, which, as already mentioned, is an 'average' approach (see text box). The new POCPs (Derwent *et al.*, 1998; Jenkin & Hayman, 1999) are based on a 'marginal' approach, being calculated from the 5-day-integrated incremental ozone production due to an incremental emission of the VOC on top of a basic emission scenario.
Andersson-Sköld *et al.* (1992) have calculated POCPs for a different trajectory with a lower background concentration of NO_x. POCPs have thus been calculated for two scenarios:

- a scenario with a relatively high background NO_x concentration (i.e. with a surplus of NO_x and VOC as limiting factor for ozone formation), yielding 'high-NO_x POCPs' (Derwent *et al.*, 1998); and
- a scenario with a relatively low background NO_x concentration (i.e. with a surplus of VOC and NO_x as limiting factor for ozone formation), yielding 'low-NO_x POCPs' (Andersson-Sköld *et al.*, 1992).

Methods based on fate factors
Hofstetter (1998) has developed characterisation factors based on Disability Adjusted Life Years (DALYs) for respiratory diseases due to air pollution (see also Section 4.3.7). These factors, which are used in the Eco-indicator 99 (Goedkoop & Spriensma, 1999), take account of ozone-induced respiratory disease for a number of VOCs and NO_x, based on a fate factor and the DALY for O_3. A fate factor for NO_x and for Non-Methane VOC (NMVOC) was derived using the EMEP model[1], with a fate factor (F_i) for other VOCs then being derived by multiplying the ratio between the POCP of the VOC in question and that of NMVOC by the fate factor for NMVOC (F_{NMVOC}) derived with the EMEP model:

$$F_i = \frac{POCP_i}{POCP_{NMVOC}} \times F_{NMVOC} \qquad (4.3.9.3)$$

The only difference between the characterisation values developed by Hofstetter (1998) and the POCPs discussed above is located in two constants: the DALY for O_3 and F_{NMVOC}. (Note that DALYs do not cover effects on ecosystems or crops, and that these effects should be assessed separately.)
Hofstetter (1998) has also developed characterisation factors for respiratory diseases for the main inorganic chemicals causing winter smog: particles, NO_x, CO and sulphur compounds. With this method summer and winter smog can therefore be amalgamated into one category indicator.

Methods based on Incremental Reactivity
An alternative approach to describing the potential contribution of VOCs to photo-oxidant formation is thyat based on the notion of Incremental Reactivity (IR; Carter 1994, Carter, 1997). The incremental reactivity of a VOC in a pollution scenario is defined as the change in ozone caused by adding a small amount of the VOC to the emissions in the scenario, divided by the amount of VOC added (Carter, 1994):

$$IR_i = \frac{\Delta[O_3]}{\Delta m_i} \qquad (4.3.9.4)$$

with:

IR_i	the Incremental Reactivity of substance I;
$\Delta[O_3]$	the change in ozone (g);
Δm_i	the incremental mass of VOC added (g).

IRs are calculated using a so-called 'base case scenario' (i.e. specific pollution scenario) that represents a specific ozone exceedence episode in an area of the United States. The base case scenario is subsequently adjusted, resulting in three derived scenarios and three associated IRs:

[1] European Monitoring and Evaluation Programme (Barret & Berge, 1996).

MIR scenario: Maximum Incremental Reactivity scenario, in which the NO_x emissions in the base case scenario are adjusted to yield the highest incremental reactivity of the initially present VOC mixture (high-NO_x)

MOR scenario: Maximum Ozone Reactivity scenario, in which the NO_x emissions in the base case scenario are adjusted to yield the highest peak ozone concentration (high-NO_x);

EBIR scenario: Equal Benefit Incremental Reactivity scenario, in which the NO_x emissions in the base case scenario are adjusted such that VOC and NOX reductions are equally effective in reducing O_3 (medium-NO_x)

These three scenarios provide three different IRs: the MIR scenario yields MIRs (Maximum Incremental Reactivity), the MOR scenario MOIRs (Maximum Ozone Incremental Reactivity) and the EBIR scenario EBIRs (Equal Benefit Incremental Reactivity (Carter, 1994)

There are three main differences between the approach based on IRs and that employing POCPs. First, while recent POCPs were developed on the basis of regional European scenarios (Derwent *et al.*, 1998; Andersson-Sköld *et al.*, 1992), IRs are grounded in scenarios for urban areas in North America (Carter 1994, Carter, 1997). Second, POCPs are based on a trajectory model of VOC transport over Europe, IRs on a single-cell box model. Third. POCPs are based on a time span of 5 days, IRs on a time span of at most one day. In other respects, too, the scenarios used for the respective calculations are at variance and are therefore difficult to compare.

At the same time, though, there is a reasonable correlation between the POCP and MIR values, which generally predict the same relative importance of different classes of VOC (Hahn & Will, 1995; Jenkin & Hayman, 1999). MIRs consider ozone formation on a much shorter time scale, however, and therefore give greater weight to those VOCs that are rapidly oxidised, whereas POCPs provide better resolution of less reactive VOCs (Jenkin & Hayman, 1999).

Moving back to the wider discussion of photo-oxidant formation, several important issues have been raised in the recent literature.

In the first place, a number of authors consider it preferable to characterise NO_x emissions under the heading of photo-oxidant formation and several attempts have been made to do so. Frischknecht (1998) bases a POCP for NO_x on the assumption that NO_x and VOCs are equally responsible for photo-oxidant formation. He then uses the quotient of annual Swiss emissions of NO_x and non-methane VOCs (NMVOCs) to calculate a POCP of 0.645 kg ethylene-equivalents per kg NO_x. Apart from this being a very crude assumption, these calculations in fact appear to be incorrect[1]. Hoffstetter (1998) calculates a fate factor for NO_x from data generated by the aforementioned EMEP model (Barret & Berge, 1996). This factor, which he himself characterises as very uncertain, is equal to that of all NMVOC combined (kg ethylene-equivalents). Finnveden *et al.* (1992), Lindfors (1996) and Nichols *et al.* (1996) propose dividing the category into two, one for NO_x and one for VOCs (and CO). Lindfors *et al.* (1995c) even add a separate subcategory for CO. However, they do not elaborate these proposals into characterisation factors for NO_x. Derwent *et al.* have calculated POCPs voor NO and NO_2.

It has also been proposed that POCPs should better reflect the fact that differences in background NO_x concentration lead to different relative contributions of individual VOCs to ozone formation. To tackle this problem, Nichols *et al.* (1996) and Hauschild & Wenzel (1998) propose using two different lists of POCPs for VOCs: one for situations with a high background NO_x concentration (data from Derwent & Jenkins, 1990, recently updated in Derwent *et al.* 1998) and one for situations with a low background NO_x concentration (data from Andersson-Sköld *et al.*, 1992). Another option would be to use MOIRs and EBIRs in situations with high or low NO_x concentrations, respectively.

Many authors hold that a distinction should be made between long-term and short-term processes (e.g. Finlayson-Pits & Pits, 1993; Nichols *et al.,* 1996). Short-term smog episodes characterised by high ozone concentrations (e.g. >120 ppb) are best understood in terms of the effects of oxidant formation on human health. However, longer periods with lower ozone levels may also cause damage to human health, crops and forest ecosystems. Both POCPs and MIRs (MOIRS and EBIRS) are concerned

[1] Frischknecht assumes that all VOC is ethylene and probably confused these numbers with those for emissions of NMVOC and NO_x

specifically with ozone formation potential in peak situations and it is unclear whether these factors are also valid for extended periods of low ozone concentration.

Finally inventory data are often insufficiently differentiated, mentioning only 'total VOC emission', or similar. One way of tackling this problem is to use average POCPs for the total VOC emissions of different types of emission source. (For a description of other approaches to the problem of group parameters, see Section 3.6) This approach has been followed by Hauschild & Wenzel (1998) and by Derwent *et al.* (1996). A second option is to solve the problem in the inventory phase, using source-specific emission profiles sources (see Section 3.6)

PROSPECTS

In the future, further progress is anticipated on Impact assessment modeling to the various endpoints, viz. damage to human health, damage to ecosystem health, etc. (*cf.* Figure 4.2.2); see Section 4.2 for a more extensive discussion of this subject).

CONCLUSIONS

As a baseline it is recommended to use the most recent POCPs of Derwent *et al.* (1998) and Jenkin & Hayman (1999), supplemented with the POCPs for inorganic substances[1] of Derwent *et al.* (1996), which include NO and NO_2.

Although MIRs, MOIRS and EBIRs also focus on ozone formation potential, these indicators consider ozone formation over a much shorter time scale. Because the longer term is relevant in the LCA context, POCPs are to be preferred over MIRs (criterion 7). The low-NO_x POCPs of Andersson-sköld *et al.* (1992) are based on the same model as the high-NO_x POCPs of Derwent *et al.* (1998) and Jenkin & Hayman (1999). Hauschild & Wenzel (1998) advise using the low-NO_x POCPs for Scandinavia and other situations with low NO_x concentrations and the high-NO_x POCPs for the rest of Europe and other regions with high NO_x concentrations. Given the fact that the latter are more frequently updated, however, the difference between the two appears to be accrue not only from regional atmospheric variation, but also from growing scientific knowledge. In our view, therefore, it is inappropriate to use both types of POCP in one and the same LCA. As a baseline we recommend using the high-NO_x POCPs, because these are more up to date and because we anticipate that most emissions covered by LCAs will occur in regions with high NO_x concentrations.

It is advised to use emission profiles to disaggregate aggregated VOC emissions before Impact assessment, rather than using POCPs for specific VOC mixtures (see Section 3.6 and Part 2b, Section 3.6). Such disaggregation is also relevant for other impact categories like human toxicity and ecotoxicity.

The evaluation of the baseline characterisation method using the characterisation factor high NOx POCP with respect to the (ISO-based) criteria is shown in Table 4.3.9.1.

Table 4.3.9.1: Evaluation of the baseline characterisation method for photo-oxidant formation, using the characterisation factor high NOx POCP, with respect to the (ISO-based) criteria of Table 4.3.1.

	criterion	evaluation
1.	scientifically and technically valid	yes; based on a widely supported trajectory model
2.	environmentally relevant	yes; the adverse effects of high ozone concentrations (especially those on human health) are well understood
3.	internationally accepted	not officially, but the POCPs of Derwent *et al*. (1998) are widely used all over the world
4.	value-choices and assumptions	yes
5.	focal point in environmental mechanism	midpoint
6.	linear	yes
7.	time span	5 days

[1] When consulted on this topic, D. Derwent confirmed that the old (average) POCPs for inorganic substances can be combined with the new (marginal) POCPs for VOCs.

	criterion	evaluation
8.	fate, exposure/intake and effects	fate included, exposure/intake not relevant, effects included in terms of effects on photo-oxidant formation
9.	less is better	yes; below-threshold effects are also included
10.	time- and location-independent	yes, although in fact POCPs are representative for European countries with high background NO_x concentrations
11.	operational	yes, for 120 VOCs
12.	uncertainty margins	it is not known how the uncertainties in high-NO_x POCPs compare with those associated with other methods for this impact category

The following methods are included in the guide as options for sensitivity analysis:
- MIRs, MOIRs and EBIRs (Carter, 1997), when there is interest in assessing VOCs on a shorter time scale.
- The low-NO_x POCPs of Andersson-Sköld *et al.* (1992) for LCA studies in which most emissions take place in regions with low NO_x concentrations.

No additional recommendations are provided for extended LCAs.

method status	characterisation method/factor	reference
baseline	high NO_x POCP	Derwent *et al*., 1996; Derwent *et al.,* 1998; Jenkin & Hayman, 1999
alternative 1	MIR	Carter, 1997
alternative 2	MOIR	Carter, 1997
alternative 3	EBIR	Carter, 1997
alternative 4	low NO_x POCP	Andersson-Sköld *et al*., 1992
additional	–	–
variant	–	–

RESEARCH RECOMMENDATIONS

Long-term research:
- The old POCPs are based on an average approach, the new POCPs on a marginal approach. This makes them a good starting point for a general analysis of the differences between using a marginal and an average approach for the same impact category.

4.3.10 Acidification

TOPIC

Acidifying pollutants have a wide variety of impacts on soil, groundwater, surface waters, biological organisms, ecosystems and materials (buildings). Examples include fish mortality in Scandinavian lakes, forest decline and the crumbling of building materials. The major acidifying pollutants are SO_2, NO_x and NH_x Areas of protection are the natural environment, the man-made environment, human health and natural resources (see Figure 4.2.2).

DEVELOPMENTS IN THE LAST DECADE

Acidification is one of the impact categories in which local sensitivity plays an important role and the possibility of including regional differences in the LCA model has been a key issue in recent years. It is therefore discussed here as an extension of the general discussion on regionalisation earlier in this chapter (see discussion on criteria below Table 4.3.1).

Heijungs *et al.* (1992)

In Heijungs *et al.* (1992) *acidification potentials* (APs) were used as a characterisation factor to assess and aggregate the interventions for the impact category acidification:

$$Acidification = \sum_i AP_i \times m_i \qquad (4.3.10.1)$$

where m_i (kg) is the mass of substance *i* released, AP_i the Acidification Potential of the substance and *Acidification* is the indicator result, which is expressed in kg SO_2-equivalents.

Potential acid deposition can be expressed in terms of potential H^+-equivalents, allowing potentially acidifying emissions to be aggregated on the basis of their capacity to form H^+ ions. It is assumed, for instance, that one molecule of SO_2 yields two H^+ ions, with one molecule of nitrogen oxides (NO_x) and one molecule of reduced nitrogen compound (NH_x) each yielding a single H^+ ion. The *acidification potential* (AP) of substance *i* is defined as the number of H^+ ions produced per kg substance relative to SO_2:

$$AP_i = \frac{h_i}{h_{SO_2}} \qquad (4.3.10.2)$$

where $?_i$ (mol·kg^{-1}) represents the number of H^+ ions that can potentially be produced per kg substance *i*, and $?_{SO2}$ (mol·kg^{-1}) the number of H^+ ions produced per kg SO_2. The emission-effect curve is a straight line through zero.

As stated, APs reflect the maximum acidification potential of a substance. The actual impact will be governed by local processes and circumstances, and will be reduced as mineralisation and denitrification rates increase. The acidification caused by a particular substance may also be reduced if the anions accompanying the hydrogen ions become bound to the impacted system (for a certain period, for it is not an infinite buffer) or absorbed and removed by biomass (Hauschild & Wenzel, 1998; Lindfors, 1996). This is particularly relevant for NO_x and NH_3, where actual acidification may vary between 0% and 100% of the potential value.

Several methods have been proposed to deal with local differences in sensitivity to acidification:

1. neglecting emissions in non-sensitive areas (e.g. Hogan *et al.*, 1996);
2. weighting emissions according to the sensitivity of the area in which they are emitted (e.g. Hauschild & Wenzel, 1998).
3. assessing a maximum and a minimum scenario (e.g. Lindfors, 1996; Nichols *et al.*, 1996);
4. extending models to include regional sensitivity and fate (e.g. Potting *et al,* 1998; Huijbregts, 1999b) .

These four options are considered in turn.

1. Neglecting emissions in non-sensitive areas

The simplest solution is to neglect all emissions occurring in non-sensitive areas, as exemplified in the method presented by Hogan *et al.* (1996). As this requires knowledge of the geographical location of the acidifying emissions (i.e. the country of origin), additional information must be collected in the Inventory phase.

2. Weighting emissions according to local sensitivity

A more sophisticated method is to weight emissions according to the sensitivity of the area in which they are emitted. Examples of this approach are Blau & Seneviratne (1995), Tolle (1997) and Hauschild & Wenzel (1998). Blau & Seneviratne (1995) propose three different sensitivity classes for Europe and for the World, while Tolle (1997) proposes state-specific scaling factors for the USA. Hauschild & Wenzel (1998) propose the same characterisation factors as Heijungs *et al.* (1992). In addition, they propose using a 'site factor' to account for local circumstances that reduce the acidifying impact of certain substances in particular areas. For instance, they report that 25% of the annual nitrogen input to forest ecosystems in Denmark is removed in harvested trees. This leads to a site factor of 0.75 for NO_x and NH_3 in Danish forests. These methods also require the location of the acidifying emissions to be

known (Blau & Seneviratne: more or less at national level; Tolle: at state level; Hauschild & Wenzel: at ecosystem level, e.g. forest, natural area[1]).

3. Assessing a maximum and a minimum scenario

Lindfors (1996) and Nichols *et al.* (1996) propose an entirely different approach. They suggest that two acidification scenarios should be studied: a maximum scenario including the contribution of NO_x and NH_3 and a minimum scenario excluding the contribution of these compounds, which may vary widely depending on soil conditions (influencing anion leaching) and ecosystem management (biomass removal).

4. Modeling regional sensitivity and fate

The above methods all take account of the sensitivity of the region in which the acidifying substances are emitted but not for the subsequent fate of the substance. Potting *et al.* (1998) propose an approach that includes both fate and regional sensitivity based on a dispersion model developed by EMEP, the Co-operative Program for Monitoring and Evaluation of the long-range transmission of air pollutants in Europe (Amann *et al.*, 1996; Barret *et al.*, 1996) and on the acidification model RAINS (Posch *et al.*, 1997):

$$I = \sum_{e \in \text{Europe}} A_{e \in j} \times \Theta(D_j - CL_{e \in j}) \qquad (4.3.10.3)$$

with

$$D_j = \sum_{x \in Acid} \sum_{i \in \text{Europe}} t_{r,x,j} \times E_{r,x} \qquad (4.3.10.4)$$

with

I	the total acidification;
$A_{e \in j}$	ecosystem e in grid cell j;
$CL_{e \in j}$	the critical load for ecosystem e in grid cell j;
D_j	the deposition on grid cell j;
Θ	a step function which is 0 if the deposition is below the CL of ecosystem e and 1 if it is above;
$t_{r,x,j}$	a transport factor: the fraction of $E_{r,x}$ deposited on *j*;
$E_{r,x}$	the emission of substance x in region r.

Although it is an advantage that this method takes account of regional differences in sensitivity and fate, there is one concern: the use of a step function. For each type of ecosystem distinguished by RAINS a critical acidification load for sulphur and nitrogen is calculated based on ecosystem properties. Europe is divided into a very large number of grid cells, and for each of these a cumulative distribution can be made of the critical loads of all the ecosystems it comprises; see Figure 4.3.10.1[2]. This step function is used to find the change in unprotected ecosystem area as a result of a change in deposition (which is in turn the result of a change in acidifying emissions).

[1] As emissions do not frequently occur in such natural areas, however, this ecosystem-based method appears to require a fate component.

[2] In reality this is rather more complicated because Rains-Europe now uses critical load functions represented by isolines for combined sulphur and nitrogen loads. However, the principle of calculating critical load exceedance remains the same.

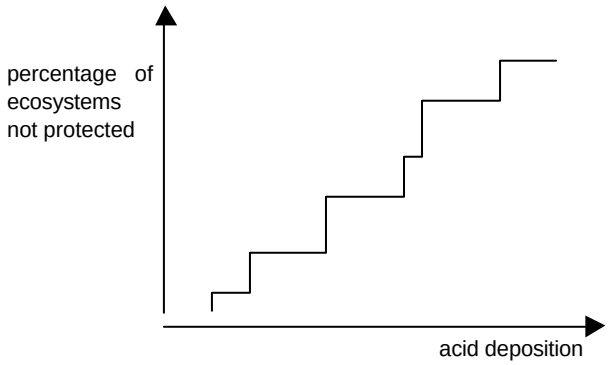


Figure 4.3.10.1: Simplified view of the cumulative distribution of critical loads in a grid cell in RAINS.

Because of this step function a marginal increase in emission will not generally lead to a change in the ecosystem area that is unprotected, although occasionally there will be a relatively large change. In other words, the derivative of A_s is either zero or infinity. This is certainly not a suitable basis for a characterisation factor, which is designed to describe the effects of marginal changes in emissions. This becomes less problematical if the number of kinds of ecosystem in a grid cell is sufficiently large and a sufficiently large 'marginal' increase in emissions is used to derive acidification factors. Potting therefore takes a 10% increase in regional emission as a 'marginal' change, which he claims is sufficient to overcome this problem. If a smaller change had been chosen, however, the resulting characterisation factors would have been different. The equivalency factors are thus highly dependent on the choice of 'marginal' emission change.

Huijbregts (1999b) has therefore proposed an improvement. He suggests using not the change in unprotected ecosystem area as the effect, but the change in relative risk. This relative risk is characterised by the ratio between deposition and critical load. Huijbregts thus replaces:

$$I = \sum_{e \in \text{Europe}} A_{e \in j} \times \Theta(D_j - CL_{e \in j})$$

by

$$I = \sum_{e \in \text{Europe}} A_{e \in j} \times \frac{D_j}{CL_{e \in j}} \qquad (4.3.10.5)$$

This solves the problem by changing the step function into a continuous function, with the characterisation factor for acidification now based on a deposition/critical load ratio simular to the PEC/PNEC ratio used for ecotoxicity. Based on this relative risk and some other minor changes to RAINS, Huijbregts has adapted RAINS to the purposes of LCA, calling it RAINS-LCA, and using it to calculate regional characterisation factors for acidification.
A second concern is the need for global characterisation factors in LCA. Potting *et al.* (1998) have calculated regional characterisation factors for approximately 40 regions in Europe, but no average European or global factors. Huijbregts (1999b) has calculated average European factors, by weighted summation of the regional factors for each acidifying substance:

$$AP_{x,r} = \frac{\sum_{e \in \text{Europe}} A_{e \in j} \times \frac{t_{r,x,j}}{CL_{e \in j}}}{\sum_{e \in \text{Europe}} A_{e \in j} \times \frac{t_{r,ref,j}}{CL_{e \in j}}} \qquad (4.3.10.6)$$

with:

$AP_{x,r}$	the regional acidification potential of substance x in region r;
A_e	ecosystem e (in grid cell j);
$CL_{e \hat{I} j}$	the critical load for ecosystem e (in grid cell j);
$t_{r,x,j}$	a transport factor: the fraction of $E_{r,x}$ deposited on *j*;
$E_{r,x}$	the emission of substance x in region r.

In the ExternE project (EC, 1995a) three different types of effects of acid rain are modeled separately: effects on ecosystems, effects on buildings, and effects on recreational fishery. Substance fate is also modeled using two different models: a Gaussian plume model and a trajectory model. However, ExternE does not provide straightforward characterisation factors that can be used directly in LCA.

For the Eco-indicator 99 (see Section 4.2), damage factors were calculated using the so-called 'Nature Planner', a fate and effect model developed for the Netherlands (Goedkoop & Spriensma, 1999). The authors have expressed doubts as to whether the model is still valid at the European or global scale, however.

PROSPECTS

In the future, further progress is anticipated on Impact assessment modeling to the various endpoints, viz. damage to human health, damage to ecosystem health, etc. (*cf.* Figure 4.2.2); see Section 4.2 for a more extensive discussion of this subject).

CONCLUSIONS

Concluding, the average European characterisation factors of Huijbregts (1999b) are recommended for the time being as the best available practice. Regional factors have not been adopted as the baseline, because it is not always possible, nor desirable, to consider differences between emission sites in LCA. It is therefore important that emission site-*in*dependent characterisation factors become available, even for those impact categories for which local sensitivity is important (criterion 10 for selection of baseline characterisation method). The average European factors of Huijbregts (1999b) are better linked to the category endpoint natural environment than the old factors of Heijungs *et al.* (1992), even when used together with first three options above: 'maximum and minimum scenarios for NO_x and NH_3' (Lindfors, 1996; Nichols *et al.*, 1996), 'neglecting emissions in non-sensitive areas' (Hogan *et al.,* 1996) and 'weighting of emissions according to the sensitivity of the area in which they are emitted' (Hauschild & Wenzel, 1998), because they account for the fate and sensitivity of the receiving ecosystem (criteria 2 and 8). The ExternE method is not operational in terms of characterisation factors (criterion 11) and the method of Goedkoop & Spriensma has an important source of uncertainty (criterion 12): "the very crude assumption that the average sensitivity of Dutch natural areas is representative for the average sensitivity of other natural areas" (Goedkoop & Spriensma, 1999).
The baseline characterisation method for acidification, using the AP based on RAINS-LCA, is evaluated with respect to the (ISO-based) criteria in Table 4.3.10.1.

Table 4.3.10.1: Evaluation of the baseline characterisation method for acidification, using the AP based on RAINS-LCA, with respect to the (ISO-based) criteria of Table 4.3.1.

	criterion	evaluation
1.	scientifically and technically valid	yes, based on the RAINS model, supported by United Nations Economic Commission for Europe (UN/ECE)
2.	environmentally relevant	yes, near-endpoint level
3.	internationally accepted	not officially, but RAINS is supported by UN/ECE
4.	value-choices and assumptions	present, but accepted by an international community
5.	focal point in environmental mechanism	near endpoint
6.	linearity	yes
7.	time span	eternity
8.	fate, exposure and effects	all included
9.	less is better	yes, effects below threshold are also included
10.	time- and location-independent	yes, although in fact they are representative for Europe
11.	operational	yes, for the three main acidifying substances: NO_x, NH_3 and SO_4
12.	uncertainty margins	greater than for the method using the APs of Heijungs *et al*. (1992)

The following method is included in the guide as an option for sensitivity analysis:

- The method using the acidification potentials of Heijungs *et al.* (1992). Several substances have been added to the 1992 list (source: Hauschild & Wenzel, 1998). This data set is useful when substances other than NH_3, SO_2 or NO_x are involved.

Recommendations for extended LCAs:

– If most emissions take place in Europe, the regional characterisation factors of Huijbregts (1999b) should be used. This means gathering extra information in the Inventory phase on the region of origin of the acidifying emissions.

method status	characterisation method/factor	reference
baseline	average European AP	Huijbregts, 1999b
alternative	generic AP	Heijungs *et al*. , 1992 (updated with Hauschild & Wenzel, 1998)
additional	region (site) dependent AP	Huijbregts, 1999b
variant	–	Hogan *et al*., 1996; regional factors of Hauschild & Wenzel, 1998; Potting *et al*.(1998)

RESEARCH RECOMMENDATIONS

Long-term research:

– It is desirable to develop a standard method to account for fate and regional differences for *all* impact categories. To this end a research programme should be initiated focusing on all impact categories for which fate and regionalisation may be important, including acidification.

4.3.11 Eutrophication

TOPIC

Eutrophication covers all potential impacts of excessively high environmental levels of macronutrients, the most important of which are nitrogen (N) and phosphorus (P). Nutrient enrichment may cause an undesirable shift in species composition and elevated biomass production in both aquatic and terrestrial ecosystems. In addition, high nutrient concentrations may also render surface waters unacceptable as a source of drinking water. In aquatic ecosystems increased biomass production may lead to a depressed oxygen levels, because of the additional consumption of oxygen in biomass decomposition (measured as BOD, biological oxygen demand). As emissions of degradable organic matter have a similar impact, such emissions are also treated under the impact category 'eutrophication'. The areas of protection are the natural environment, natural resources and the man-made environment (see Figure 4.2.2).

SETAC-Europe has also assigned the emission of waste heat to this impact category. In this guide, however, waste heat is treated as a separate category (see Section 4.3.12).

DEVELOPMENTS IN THE LAST DECADE

Hauschild & Wenzel (1998) take more or less the same approach as Heijungs (1992). They calculate nutrient enrichment potentials relative to NO_3^- this being one of the key nutrifying agents. They also base these potentials on the assumption that one mole of P contributes as much to the formation of biomass as 16 moles of N. Besides this overall nutrient enrichment potential they also propose two separate potentials for nitrogen and phosphorus, expressed as gram N and gram P per gram substance, which can be used if it is known on the basis of site-dependent information which of the nutrients is limiting. Only biologically available nitrogen is considered (N_2 is not classified). Hauschild & Wenzel (1998) do not assign BOD or waste heat to the impact category eutrophication.

Heijungs *et al.* (1992)

In Heijungs *et al.* (1992) the eutrophication (there termed 'nutrification') due to N, P and C (measured in terms of Chemical Oxygen Demand, COD) was aggregated by quantifying their potential contribution to biomass formation. *Eutrophication potentials* (EPs) were used as a characterisation factor to assess and aggregate the interventions for the impact category eutrophication:

$$Eutrophication = \sum_i EP_i \times m_i \qquad (4.3.11.1)$$

where m_i (kg) is the mass of substance *i* released to the air, water or soil, EP_i the Eutrophication Potential of the substance and *Eutrophication* the indicator result, which is expressed in kg PO_4^{3-}-equivalents. EP_i (dimensionless) reflects a substance's potential contribution to biomass formation. As a formula:

$$EP_i = \frac{n_i / M_i}{n_{ref} / M_{ref}} \qquad (4.3.11.2)$$

where $?_i$ and $?_{ref}$ are the potential contributions to eutrophication of one mole of substance *i* and *ref* (i.e. PO_4^{3-}), respectively, and M_i and M_{ref} (kg·mol^{-1}) are the mass of *i* and *ref* (i.e. PO_4^{3-}). EPs are based on the 'average' chemical composition of aquatic organisms: $C_{106}H_{263}O_{110}N_{16}P$ (Stumm & Morgan, 1981), assumed to be representative of average biomass. The only emissions considered to be eutrophying by Heijungs *et al.* (1992) were emissions of nitrogen and phosphorus compounds. They determined the contribution of each of these nutrients to biomass formation, assuming unlimited supply of other nutrients. In this approach one mole of biomass requires 16 moles of N and 1 mole of P. When degradable organic matter is emitted, the O_2 required for its degradation can be measured as chemical oxygen demand (COD). It is assumed that degradation of one mole of biomass requires 138 moles of O_2. Therefore if the contribution to eutrophication, $?_P$, of one mole of P is 1, the contribution of one mole of N, $?_N$, is 1/16 and the contribution of COD (as O_2), $?_{COD}$, is 1/138. The contribution of one mole is then expressed as the contribution of one gram by dividing by the molecular weight. In the formula for EP_i, $?_i$ is the total contribution of one mole of substance *i* and M_i is the molecular weight of *i.* The reference substance PO_4^{3-} is used to create *eutrophication potentials* (EPs), similar to ODPs, GWPs, APs and POCPs.

Hence, the characterisation factor for eutrophication is independent of whatever substance happens to be the limiting factor in a particular location. This approach was adopted for two reasons: to obtain universal, i.e. global characterisation factors, independent of local differences, and because it is unknown which medium (freshwater, salt water, groundwater or soil) an emitted substance will eventually enter.

A general comment in the literature is that the approach of Heijungs *et al.* (1992) disregards the media of emission as well as the sensitivity of the receiving environment and the limiting nutrient. Two suggestions have been made for overcoming these limitations: by distinguishing ecosystem subcategories, and by including fate and site- or region-dependent effect modeling.

The first solution, proposed by Nichols *et al.* (1996) and by Lindfors *et al.* (1995c), involves distinguishing ecosystem subcategories. Nichols *et al.* distinguish 3 subcategories:

i) terrestrial ecosystems,
ii) aquatic ecosystems: smaller inland surface waters, and
iii) aquatic ecosystems: larger inland surface waters and seas.

Lindfors *et al.* (1995c) propose 5 subcategories representing 5 scenarios:

1. terrestrial ecosystems, emissions of N to air (because most terrestrial ecosystems are N-limited)
2. aquatic ecosystems, P-limited; emissions of P and organic matter to water
3. aquatic ecosystems, N-limited; emissions of N and organic matter to water
4. aquatic ecosystems, N-limited; emissions of N to air and water and organic matter to water
5. aquatic ecosystems, emissions of N, P and organic matter to water and N and P to air.

The emissions in these scenarios can be characterised in terms of either PO_4^{3-}- or NO_3^--equivalents or oxygen demand. In the latter case the characterisation factor of substance *i*, in g oxygen/kg *i*, is the oxygen required for the mineralisation of the organic matter (average composition) produced from one kg of *i* when *i* is the limiting nutrient, with one mole of N and P corresponding respectively to 8.6 and 138 moles of consumed O_2. The two types of units for the characterisation factor boil down to the same and are in fact interchangeable.

The second option suggested is to include fate and site- or region-dependent effect modeling. Besides acidification factors, Huijbregts (1999b) calculated eutrophication factors for air emissions of NH_3 and NO_x using RAINS-LCA, which also contains critical loads for eutrophication. He calculated both regional and average European factors. These factors account only for eutrophication of terrestrial ecosystems due to air emissions. Direct emissions to soil are not included, nor are emissions to water. Effects on aquatic ecosystems are likewise not included.

For the Eco-indicator 99, damage factors were calculated using the so-called 'Nature Planner', a fate and effect model developed for the Netherlands (Goedkoop & Spriensma, 1999). The authors have expressed doubts as to whether the model is still valid at the European or global scale, however.

PROSPECTS

In the future, further progress is anticipated on Impact assessment modeling to the various endpoints, viz. damage to human health, damage to ecosystem health, etc. (*cf.* Figure 4.2.2); see Section 4.2 for a more extensive discussion of this subject).

Huijbregts & Seppälä (2000) recently proposed a method for LCA assessment of the effects of eutrofication on aquatic ecosystems. This method combines the fate part of RAINS-LCA for air emissions with the characterisation factors of Heijungs *et al.* (1992). Water emissions are also included, using a fate factor of one (i.e. all emissions to water remain in water).

CONCLUSIONS

Concluding, our preference would be for assessment of eutrophication using a method that includes multi-media fate and exposure as well as the varying sensitivity of the ecosystems exposed. At present, however, such a method is only available for the terrestrial effects of air emissions of NH_3 and NO_x (criterion 11). Huijbregts & Seppälä (2000) propose using this method for terrestrial eutrophication in conjunction with a separate method for aquatic eutrophication, combining a fate factor (calculated with RAINS-LCA for air emissions of NH_3 and NO_x and set to 1 for water emissions) with the old characterisation factors of Heijungs *et al.* (1992). However, this proposal has two drawbacks: i) emissions to soil are not included, although these may be particularly relevant in LCA studies covering agricultural products (criterion 11), and ii) using different methods for terrestial and aquatic eutrophication would yield two subcategories that cannot simply be summed. The environmental profile would thus contain two scores instead of one for eutrophication. The method of Goedkoop & Spriensma has an important source of uncertainty (criterion 12): "the very crude assumption that the average sensitivity of Dutch natural areas is representative for the average sensitivity of other natural areas" (Goedkoop & Spriensma, 1999).

We therefore propose adopting as a baseline the method described in Heijungs *et al.* (1992), in which all emissions of N and P to air, water and soil and of organic matter to water are aggregated into a single measure, because this method allows both terrestrial and aquatic eutrophication to be assessed. The characterisation factors in PO_4^{3-}-equivalents, NO_3^-- equivalents and O_2-equivalents are all interchangeable, and in this Guide we opt to use the same concept and factors as Heijungs *et al.*, in PO_4^{3-}-equivalents. Several substances have been added to the original 1992 list.

The baseline characterisation method for eutrophication, using the eutrophication potentials of Heijungs *et al.* (1992) as a characterisation factor, is evaluated with respect to the (ISO-based) criteria adopted in this Guide in Table 4.3.11.1.

Table 4.3.11.1: Evaluation of the baseline characterisation method for eutrophication, using the eutrophication potentials of Heijungs *et al.* (1992) as a characterisation factor, with respect to the (ISO-based) criteria of Table 4.3.1.

	criterion	evaluation
1	scientifically and technically valid	not relevant
2	environmentally relevant	category indicator is close to intervention
3	internationally accepted	no, but widely used
4	value-choices and assumptions	present (e.g. fixed C÷N÷P ratio is representative for all biomass, terrestrial and aquatic)
5	focal point in environmental mechanism	midpoint
6	linearity	yes
7.	time span	eternity
8	fate, exposure and effects	fate and exposure not included
9	less is better	yes, effects below threshold also included
10	time- and location-independent	yes
11.	operational	yes, for many substances containing bioavailable N or P and COD
12	uncertainty margins	low

The following method is included in the Guide as an option for sensitivity analysis:
- The method using the average European eutrophication potentials for terrestrial ecosystems (Huijbregts, 1999b).

Recommendations for extended LCAs:
- If most emissions take place in Europe, the regional characterisation factors of Huijbregts (1999b) should be used. This means gathering extra information in the Inventory phase on the region of origin of the eutrophying emissions.
- If there are compounds containing bioavailable N or P that are relevant for the study but are not included in the list, calculate characterisation factors using the formula specified above.

method status	characterisation method/factor	reference
baseline	generic EP	Heijungs *et al.*, 1992 (updated)
alternative	average European EP	Huijbregts, 1999b
additional	region (site) dependent EP	Huijbregts, 1999b
variant	–	

RESEARCH RECOMMENDATIONS

Long-term research:
- It is desirable to develop a standard method to account for fate and regional differences for *all* impact categories. To this end a research programme should be initiated focusing on all impact categories for which fate and regionalisation may be important, including eutrophication.

4.3.12 Waste heat

TOPIC

Emissions of waste heat may increase temperatures on a local scale: in a city or lake, for example. They cannot contribute to global warming on a scale such as that associated with emissions of greenhouse gases. The effects on ecosystems of waste heat emissions to the air are negligible. Depending on local conditions, the discharge of waste heat into surface waters may result in a

substantial temperature rise, with a consequent impact on local aquatic ecosystems. SETAC-Europe includes waste heat under the impact category 'eutrophication', because it may likewise lead to lower oxygen concentrations (measured as COD, for example). In this Guide waste heat is treated as a separate impact category, although it covers only aquatic emissions of waste heat such as cooling water emissions from power stations. The areas of protection are the natural environment and natural resources.

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)
Waste heat is treated as a separate impact category in Heijungs *et al.* (1992). The local character of the problem is abstracted by expressing waste heat emissions into water in megajoules (MJ). These emissions can be quantified by estimating the proportions of waste heat discharged via the cooling water and via the air (based on the energy balance of the plant). As this data is available from the Inventory analysis, no further action is required in the classification step. In essence, the characterisation factor for all waste heat emissions to water is 1, and 0 for all waste heat emissions to the atmosphere.

$$Aquatic\ heat = E_{water} \qquad (4.3.12.1)$$

There are no new developments relating to the topic of heat emissions to surface water.

PROSPECTS
No specific developments are foreseen in this area.

CONCLUSIONS
We propose using the method described in Heijungs *et al.* (1992). This means that all heat emissions to water are multiplied by a charaterisation factor of 1. This baseline characterisation method is evaluated with respect to the (ISO-based) criteria adopted in this Guide in Table 4.3.12.1.

Table 4.3.12.1: Evaluation of the baseline characterisation method for waste heat, using characterisation factor = 1, with respect to the (ISO-based) criteria of Table 4.3.1.

	criteria	evaluation
1.	scientifically and technically valid	not relevant
2.	environmentally relevant	category indicator is close to intervention
3.	internationally accepted	no
4.	value-choices and assumptions	no
5.	focal point in environmental mechanism	close to intervention
6.	linearity	yes
7.	time span	not relevant
8.	fate, exposure and effects	fate and exposure not included
9.	less is better	yes
10.	time- and location-independent	yes
11.	operational	yes
12.	uncertainty	very low

No methods are included in this Guide as options for sensitivity analysis.
No additional recommendations are given for extendend LCAs.

method status	characterisation method/factor	reference
baseline	unweighted aggregation of energy	Heijungs *et al.,* 1992
alternative	–	–
additional	–	–
variant	–	–

RESEARCH RECOMMENDATIONS
No research is foreseen.

4.3.13 Odour

TOPIC

Odour becomes a problem when a given concentration of odorous substances is experienced as unpleasant. Whether an odour is experienced as stench will depend on the particular individual exposed. Above a certain emission level, however, every individual will experience it as such. Here, the term odour will be used for effects. The area of protection is human health.

Odour may be defined as the observed difference between a sample of clean air and a sample of contaminated air. The concentration at which such a difference cannot quite be observed varies from substance to substance, and depends on the physical and chemical properties of the substance (Brasser *et al.*, 1985). The *odour threshold value* of a substance is defined as the concentration of that substance under defined standard conditions at which 50% of a representative sample of the population can just detect the difference between a sample of air mixed with that substance and a sample of clean air. Odour can be measured fairly objectively, while odour nuisance is more a matter of individual sensitivity.

DEVELOPMENTS IN THE LAST DECADE

We are not aware of any new developments in LCIA relating to the topic of odour. Indeed, most LCIA methodologies do not even have an impact category 'odour'. Lindfors *et al.* (1995c) propose that odour be assigned to their impact categories 'human health' and 'habitat alterations'. Within the former category they suggest using the same method for odour as Heijungs *et al.* (1992). The only difference is the name of the impact category. Within the category 'habitat alteration' no category indicator has yet been operationalised for odour.

Heijungs *et al.* (1992)

In Heijungs *et al.* (1992) emissions of odorous substances were classified using a method similar to the critical volumes approach, by dividing the emission of a potentially malodorous substance by the odour threshold value of that substance. A distinction must be made between emissions of potentially malodorous substances to the atmosphere and to water, for each is associated with a different odour threshold value, as expressed in the following formulae:

$$Malodourous\ air = \sum_i \frac{m_{i,air}}{OTV_{i,air}} \qquad (4.3.13.1)$$

$$Malodourous\ water = \sum_i \frac{m_{i,water}}{OTV_{i,water}} \qquad (4.3.13.2)$$

where *malodorous air* is the quantity of air contaminated to the odour threshold value (m^3), $m_{i,\,air}$ the emission of substance *i* into the atmosphere (kg) and $OTV_{i,\,air}$ the odour threshold value in air of substance *i* ($kg \cdot m^{-3}$), and where *malodourous water* $m_{i,\,water}$ and $OTV_{i,\,water}$ represent the same parameters for water.

This provisional, critical volumes approach does not account for processes of dispersal, degradation or transformation, which are highly dependent on the particular substance involved. This also means there is no assessment of indirect odour emissions such as ozone. Direct ozone emissions are negligible, particularly in comparison with the ozone formed by photochemical reactions involving volatile organic compounds and NO_x (see also Section 4.3.9).

This provisional approach was only partially adopted in the old Guide, the main reason being that no uniform odour threshold values had yet been agreed for many substances. For atmospheric emissions a comprehensive list of odour thresholds was calculated, however, using a uniform method. No such list was yet available for water.

PROSPECTS

Because odour is not generally considered to be as significant a category as, say, human health or climate change, the proposal of Lindfors *et al.* (1995c) to include odour in an impact category 'human health' seems reasonable. If and when such an impact category is distinguished aggregating all human health impacts, in the form of DALYs (see Section 4.2) for example, odour might be assigned to this new category.

Until then, it should be possible to develop *Odour potentials* (OPs) in the same way as the *human toxicity potentials* and *aquatic ecotoxicity potentials* developed by Guinée *et al.* (1996) or Huijbregts (1999a). A multi-media model might then be used to calculate the fate of the substance and the resulting environmental concentrations divided by the odour threshold. The resulting values could then, finally, be compared with the corresponding values for a reference substance.

Although the analogy between OPs and HTPs may be an advantage if odour is assigned to human health, the effects on which the threshold values are based are nonetheless very different: adverse health effects for HTPs and annoyance or maybe headaches for OPs.

CONCLUSIONS

For the present it is recommended to use the critical volumes method described by Heijungs *et al.* (1992) for malodorous air. No method is yet available for malodorous water. It is proposed to include fate at some time in the future, possibly using a method analogous to that developed for toxic substances by Guinée *et al.* (1996) or by Huijbregts (1999a). The score of this approach with respect to the (ISO) requirements is shown in Table 4.3.13.1.

Table 4.3.13.1: Evaluation of the baseline category indicator for malodorous air, the critical volumes approach of Heijungs *et al.* (1992), with respect to the (ISO-based) criteria of Table 4.3.1.

	criteria	evaluation
1	scientifically and technically valid	no
2	environmentally relevant	more or less; odour is not the same as odour nuisance, which may affect human well-being
3	internationally accepted	no
4	value-choices and assumptions	by adoptng the odour threshold level as a reference, a value choice is made to value every odour as malodorous
5	focal point in environmental mechanism	midpoint
6	linearity	yes
7	time span	not relevant
8	fate, exposure and effects	fate and exposure not included
9	less is better	yes, effects below threshold are also included
10	time- and location-independent	yes
11	operational	yes
12	uncertainty margins	unknown

No methods are included in this Guide as options for sensitivity analysis.

Recommendations for extended LCAs:
Develop OPs by using USES-LCA for the fate part and 1/OTV for the effect part.

Malodorous air

method status	characterisation method/factor	reference
baseline	inverse OTV	Heijungs *et al.*, 1992
alternative	–	–
additional	based on fate model and odour threshold values	–
variant	–	–

Malodorous water

method status	characterisation method/factor	reference
baseline	–	–
alternative	–	–
additional	–	–
variant	–	–

RESEARCH RECOMMENDATIONS

Short-term research:

– It is recommended to investigate the scope for developing OPs that include fate. The potential for integrating odour in the categories 'human health' and possibly 'ecosystem health' should also be researched. In the case of 'human health' weighting factors might, for example, be developed to weight odour nuisance with reference to toxic effects.

4.3.14 Noise

TOPIC

Noise, or noise nuisance, refers to the environmental impacts of sound. In principle, these impacts could cover at least human health and ecosystem health, but the environmental mechanisms are complex, non-linear and highly dependent upon local circumstances. Moreover, noise is similar to odour in that a given level of exposure is experienced differently by different individuals. Something considered a nuisance by one person might be appreciated by another, as exemplified by the case of loud music. Hence, whether or not sound waves will lead to 'nuisance' depends partly on the actual situation and partly on the person interviewed.

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)
Heijungs *et al.* (1992) aggregated the sound production (in $Pa^2 \cdot s$) obtained in the inventory to yield an abstract sound level that is non-location- and non-person-specific and referred to it as "potential noise nuisance". In this case the characterisation factor is essentially 1. This ignores the fact that some sound emissions may not cause any nuisance at all (e.g. source remote from those exposed) while others may cause a great deal of nuisance in certain environments (e.g. in a road tunnel os street canyon):

$$Noise = G \qquad (4.3.14.1)$$

where G is the sound production (in $Pa^2 \cdot s$).

Most LCIA methodologies do not have an impact category 'noise'. This runs counter to the observed fact that most people deem noise to be a major environmental problem. Lindfors *et al.* (1995c) propose assigning noise to the impact categories 'human health', 'human health in the working environment' and 'habitat alterations'.

Lafleche & Sacchetto (1999) describe a methodology that attempts to include noise in LCA. This methodology embodies a regional approach which is operationalised for one specific case: transport by car or truck along the Bologna-Milan highway. This methodology is still insufficiently developed for general use in LCA.
Müller-Wenk (1999) propose a method for expressing road traffic noise in terms of DALYs (see Section 4.2). This method links transport kilometres directly to DALYs, thereby more or less skipping the inventory phase. As a consequence, this method is useful for road traffic noise only and not for aircraft noise and so on.

PROSPECTS
In the future, further progress is anticipated on Impact assessment modeling to the various endpoints, viz. damage to human health, damage to ecosystem health, etc. (*cf.* Figure 4.2.2); see Section 4.2 for a more extensive discussion of this subject). The method of Müller-Wenk (1999) might provide a starting point for endpoint modeling of noise.

CONCLUSIONS
For the present it is recommended to use the method described by Heijungs *et al.* (1992) as the baseline characterisation method, thus mutiplying all sound produced by a characterisation factor of 1. This method is evaluated with respect to the (ISO-based) criteria adopted in this Guide in Table 4.3.14.1.

Table 4.3.14.1: Evaluation of the baseline characterisation method for noise, using characterisation factor = 1, with respect to the (ISO-based) criteria of Table 4.3.1.

	criteria	evaluation
1.	scientifically and technically valid	not relevant
2.	environmentally relevant	category indicator is close to intervention
3.	internationally accepted	no
4.	value-choices and assumptions	no
5.	focal point in environmental mechanism	close to intervention
6.	linearity	yes
7.	time span	not relevant
8.	fate, exposure and effects	fate and exposure not included
9.	less is better	yes
10.	time- and location-independent	yes
11.	operational	yes,
12.	uncertainty margins	very low

No methods are included in this Guide as options for sensitivity analysis.

Recommendations for extended LCAs:
- The method of Müller-Wenk (1999) is recommended as a supplementary method for extended LCAs in which road traffic noise is an important item.

method status	characterisation method/factor	reference

baseline	unweighted aggregation of sound	Heijungs *et al.*, 1992
alternative	–	–
additional	based on DALY	Müller-Wenk, 1999
variant	–	–

RESEARCH RECOMMENDATIONS

Long-term research:

- It is recommended to investigate the scope for future assignment of noise to the categories 'human health' (in terms of DALYs) and perhaps 'ecosystem health'. It is also recommended to develop DALY-based characterisation factors for the intervention noise, or sound.

4.3.15 Impacts of ionising radiation

TOPIC

The impact category 'impacts of ionising radiation' covers the impacts arising from releases of radioactive substances as well as direct exposure to radiation, in building materials for example. Exposure to ionising radiation is harmful to both human beings and animals. The areas of protection are therefore human health, the natural environment and natural resources (see Figure 4.2.2). Ionising radiation is expressed in terms of the number of atoms disintegrating (or decaying) per unit time. The SI unit of radioactivity is the becquerel (Bq), one Bq corresponding to one disintegration per second. The radioactivity of a substance is expressed in $Bq \cdot kg^{-1}$ or $Bq \cdot l^{-1}$. Radioactivity always declines in the course of time and the time taken for the radioactivity of a given substance to decline by half is known as the half-life of the substance.

The amount of radioactive material is generally expressed in mass terms, i.e. in kg. The relationship between emitted radiation r and mass m is given by:

$$r = \frac{1000 \ln 2}{t} \times \frac{N_A}{M} \times m \qquad (4.3.15.1)$$

where r is the radiation in Bq, m the mass in kg, t the half-life in s, M the molecular mass in $g \cdot mol^{-1}$ and N_A is Avogadro's number in mol^{-1}.

Different forms of radiation may be released in the process of radioactive disintegration, viz. alpha, beta, gamma and neutron radiation and X-rays. These forms of radiation may add or remove electrons to or from the atoms they encounter, i.e. ionise them. The degree of ionisation depends on the type of radiation and the material irradiated. During ionisation of a material, energy is transferred to that material. As a result the energy of the radiation is reduced and after a number of ionisations will have been fully absorbed by the irradiated material or tissue. The energy absorbed per unit mass material or tissue is known as the absorbed dose (Jonker *et al.*, 1988).

The different types of ionising radiation impinge very differently on living tissue. The effects of a given dose of alpha radiation will differ from those resulting from the same dose of gamma radiation, for example (Jonker *et al.*, 1988). This difference in harmfulness is expressed by means of a so-called 'quality factor': 20 for alpha radiation, 1 for beta and gamma radiation and X-rays, and 10 for neutron radiation (these types of radiation are abbreviated to a, ß, ?, X and n, respectively). Multiplying the absorbed dose by this quality factor yields the 'dose equivalent', expressed in sievert (Sv). Thus, one dose equivalent of alpha radiation is assumed to have the same aggregate effect as twenty dose equivalents of gamma radiation.

It is to be noted that characterisation of the impacts of ionising radiation will remain unfeasible until such time as data on emissions of radioactive substances are included in the Inventory analysis.

DEVELOPMENTS IN THE LAST DECADE

Most LCIA-methodologies do not have an impact category 'impacts of ionising radiation'. Lindfors *et al.* (1995c) propose assigning ionising radiation to the impact categories 'human health' and 'ecotoxicological impacts'. They do not operationalise a category indicator, however.

In connection with the impact category 'impacts of ionising radiation' Heijungs (1994b) distinguishes two types of emission:

Heijungs *et al.* (1992)
Heijungs *et al.* (1992) discussed the scope for including ionising radiation as an LCA impact category. They saw scope for developing *ionisation creating potentials* (ICPS) based on models used to calculate the ratio between the absorbed and equivalent doses of a substance and the absorbed and equivalent doses of a reference substance. They also discuss assessment of the impacts of ionising radiation by means of an adapted *critical volumes* approach, by dividing emissions of a radionuclide by some form of 'radiation standard' for the substance in question. For lack of a better alternative they propose using ALIs (*Annual Limits of Intake*: defined for workplace exposure by the *International Commission on Radiological Protection* (ICRP, 1979, 1991)). It should be noted that the critical volumes approach is a worst-case approach and ignores differences between substances in terms of half-lives and absorbed doses.
However, Heijungs *et al.* (1992) do not ultimately include a characterisation method for (the impacts of) ionising radiation.

- emission of radioactive substances to the environment, with subsequent release of radiation during decay;
- direct emission of radiation to the environment, without actual emission of a radionuclide (e.g. from building materials);

and two types of radiation impact:

- impacts resulting from internal exposure to radiation, due to intake of air, food or drinking water;
- impacts resulting from external exposure to radiation, due to the presence of radioactive substances in the human environment or in ecosystems.

Internal exposure can result only from physical emissions of radionuclides. For this kind of exposure Heijungs proposes a method including fate, broadly similar to the method for human toxicity adopted by Guinée *et al* (1996). Instead of ADI (Acceptable Daily Intake) the aforementioned ALI (Annual Limit of Intake) might be used. A possible provisional method is the critical volumes approach:

$$R_{internal} = \sum_i \frac{m_i}{ALI_i} \qquad (4.3.15.2)$$

where m_i (kg) is the emission of substance i and ALI_i the Annual Limit of Intake for that substance in Bq/kg of body weight.

External exposure can result from both types of emission: emission of radionuclides or direct emission of radiation. Adequate characterisation of this kind of exposure requires inclusion in the inventory of at least three technical parameters:

- the magnitude of the source in Bq;
- the characteristic decay energy of the radionuclide concerned in eV;
- the type of particle emitted during decay (a, ß$^+$, ß$^-$, ?, n).

If all these properties are known, one can sum overall external exposure to emitted radiation and to emitted radioactive substances, as expressed in the following formula:

$$R_{external} = \sum_i Q_i \times r_i \times E_i + R_{external,emissions} \qquad (4.3.15.3)$$

where Q_i is the quality factor of type i radiation (1 for ß and ?, 0.1 for n, 0.05 for a), $?_i$ the magnitude of the source (Bq) and E_i the average decay energy of the radionuclide involved (eV; electronvolt, a unit of energy used in most tables of radionuclides).

Heijungs sees two severe limitations to this approach, however: it does not include the relationship between radiation and absorbed dose, and the assumed linear additivity of low and high energy radiation is unrealistic.

Solberg-Johansen (1998) suggests a different approach for characterising the risk to humans of emissions of radioactive substances which allows the probability of human exposure to be included. It is based on the following scheme (Figure 4.3.15.1):

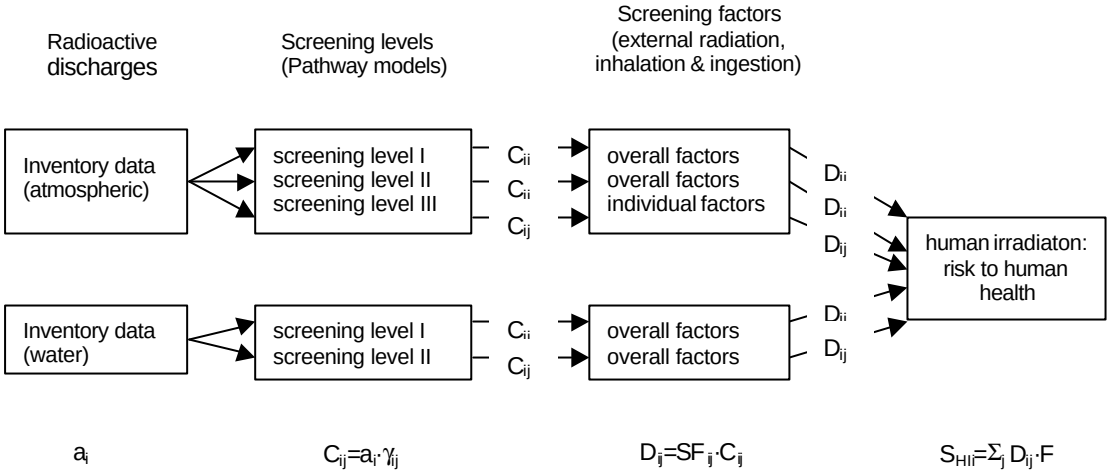


Figure 4.3.15.1: Solberg-Johansen's approach to characterising radioactive emissions (source: Solberg-Johansen, 1998).

In the figure, a_i is the activity of an emission pulse of radionuclide i (Bq), γ_{ij} a 'dispersion factor' based on the dispersion 'models' used ($y \cdot m^{-3}$), C_{ij} the 'exposure concentration' of i in medium j ($Bq \cdot y \cdot m^{-3}$) and SF_{ij} is a 'screening factor' for i in j ($Sv \cdot m^3 \cdot Bq^{-1} \cdot y^{-1}$). This screening factor is based on the exposure pathways for human beings (via food, air and external radiation) and the dose per unit intake ($Sv \cdot Bq^{-1}$). D_j then represents the dose incurred due to emission of i to compartment j (Sv). F is a probability coefficient expressing the probability of occurrence of detrimental health effects like cancer or hereditary disease (Sv^{-1}). S_{HIi} is then the contribution to human irradiation by radionuclide i, defined as the annual risk of detrimental health effects. Screening factors (SF_{ij}) and F-values for a range of radionuclides have been published by the International Commission on Radiological Protection (ICRP, 1979, 1990 and 1991).

The different screening levels in Figure 4.3.15.1 stand for different levels of complexity in the dispersion models employed. Level I embodies a very simple and conservative approach which assumes that the concentration in the receiving environment is equal to the concentration in the stream emitted:

$$radiation = \sum_j \sum_i a_i \times ?_{ij} \times SF_{ij} \times F$$

$$\text{for emissions to surface water}: g_{ij} = \frac{1}{V_j} \qquad (4.3.15.4)$$

$$\text{for emissions to air}: g_{ij} = \frac{0.25}{V_j}$$

where a_i is the activity of an emission pulse of radionuclide i (Bq), γ_{ij} the dispersion factor for radionuclide i in environmental medium j ($y \cdot m^{-3}$), SF_{ij} the screening factor for i in j ($Sv \cdot m^3 \cdot Bq^{-1} \cdot y^{-1}$), V_j the volume flow rate of the discharge to j ($m^3 \cdot y^{-1}$) and F the probability coefficient (Sv^{-1}).

Level II accounts for dispersion in the atmosphere and surface waters by using a Gaussian plume model for emissions to air and an advection-diffusion model for emissions to water. Intermedia transport is not accounted for. Level III, developed for air emissions only, uses a Gaussian plume model in combination with individual screening factors for the various exposure pathways. Levels II and III require site-specific data for the dispersion models.

Solberg-Johansen recommends using the level II procedure in LCA. However, this would mean that instead of using substance-specific characterisation factors, for each local process emitting radionuclides a specific dispersion model would have to be used to estimate the fate of these substances. A regionalised inventory is also needed, including information on weather conditions and/or conditions in the surface waters concerned. Solberg-Johansen states that the level I procedure might be used as a global approach. In this case the only 'site-specific' information required is the volume flow rate of the discharge in which the radionuclide was emitted. Both the level I and the level II procedure are operational. provided that the appropriate information is collected in the Inventory phase.

Besides the risk to human health posed by radioactive emissions, Solberg-Johansen (1998) also introduces two other topics: the human risk posed by storage of radioactive solid waste and the general risk to the environment of radionuclides. For both of these she has also developed methods, which we briefly describe.

The risk posed by radionuclides in solid waste was estimated on the basis of on a specifically British waste management scenario:

- Low Level Waste (LLW): characterisation factors based on a site-specific study on the Driggs disposal site by Smith *et al.* (1998);
- Intermediate Level Waste (ILW): characterisation factors based on a study by PAGIS (PAGIS, 1988);
- High Level Waste (HLW): characterisation factors based on a study by PACOMA (Mobbs *et al.*, 1991).

Characterisation factors are expressed in $Bq^{-1} \cdot yr^{-1}$ for LLW, and in Bq^{-1} for ILW and HLW. Inventory data should therefore be in Bq·yr or Bq per nuclide. As the method is based on the UK situation, it is probably not directly applicable to other waste management scenarios. Although characterisation factors are presented for each scenario, no full description is given of how these were calculated.

The method adopted by Solberg-Johansen to assess the environmental impacts of radionuclide emissions is based on the same fate models as those used for human impacts. It is only the effect component that differs, the factor SF_{ij} x F for human exposure in the previous equation being replaced by:

$$\frac{1}{EI_{ij} \times t_i} \qquad (4.3.15.5)$$

where EI_{ij} is the Environmental Increment factor for radionucide i in medium j, defined as one standard deviation of the mean background concentration, and τ_i is the half-life of radionuclide i. This method for calculating environmental impacts of exposure is operational for emissions of a number of radionuclides to water and soil, but not to air.

Frischknecht *et al.* (2000) have also developed a method to assess human radiation impacts (see Figure 4.3.15.2):

$$radiation = \sum_{ecomp} \sum_{i} DamageFactor_{ecomp,i} \times a_{ecomp,i} \qquad (4.3.15.6)$$

with:

radiation	the indicator result (yr);
$DamageFactor_{ecomp,i}$	the characterisation factor for substance *i* emitted to compartment *ecomp* based on DALYs ($yr \cdot kBq^{-1}$);
$a_{ecomp,i}$	the activity of substance *i* emitted to compartment *ecomp* (kBq).[1]

The method accounts for fate, exposure and effect. Fate modeling is based on EC (1995b), using a Gaussian plume model (for France) for air releases and a simple (regional) box model for water releases.

[1] We have written the unit 'yr' for the indicator result and '$yr \cdot kBq^{-1}$' for the characterisation factor, where Frischknecht *et al.* (2000) write 'DALYs' and '$DALYs \cdot kBq^{-1}$'. This is in agreement with Section 2.4, where it was concluded that SI-units are to be used. In fact, 'DALY' can be seen as the name of the quantity (like 'length'), and 'yr' as one possible unit for measurement (like 'metre').

For several globally dispersed radionuclides, global models are used in addition to the regional models. Exposure modeling is based on EC (1995b) and UNSCEAR (1993), and exposure calculated as absorbed dose. Effect modeling uses information on carcinogenic and hereditary effects (sources: Ron & Muirhead, 1998; ICRP, 1990) and the DALY approach (Murray & Lopez, 1996 and Hofstetter, 1998; *cf.* text box in Section 4.2). The Frischknecht method yields characterisation factors incorporating fate, exposure and effect. Two sets of factors are available: for a time horizon of 100,000 years, without age-weighting of DALYs, and for a horizon of 100 years, with age-weighting.

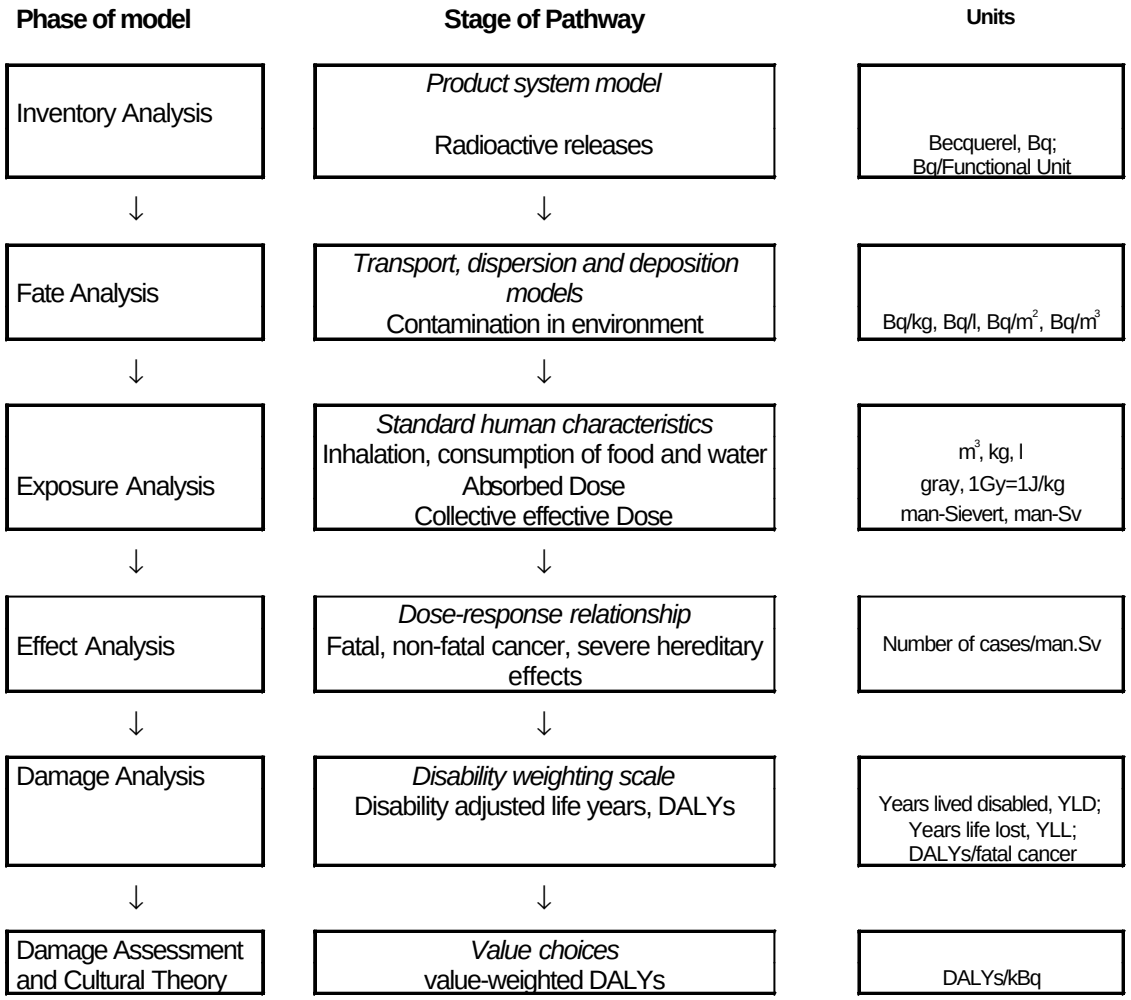


Figure 4.3.15.2: Overview of the impact pathway stages in the characterisation approach for radioactive emissions from Frischknecht *et al.* (2000).

PROSPECTS

In the future, further progress is anticipated on Impact assessment modeling to the various endpoints, viz. damage to human health, damage to ecosystem health, etc. (*cf.* Figure 4.2.2); see Section 4.2 for a more extensive discussion of this subject).

No further new developments are known to be in progress.

CONCLUSIONS

We recommend using the characterisation method of Frischknecht *et al.* (2000) as the currently best available practice for assessing the human impact of radioactive emissions to air and water. This method covers fate and exposure (criterion 8 for selection of baseline characterisation method). As a baseline we recommend using the characterisation factors without DALY age-weighting (criterion 4) based on the longest time horizon (=100,000 years; criterion 7).

The method of Solberg-Johansen (1998) is operational for more radionuclides than that of Frischknecht *et al.* (2000). However, the level I procedure of Solberg-Johansen (1998) includes only exposure and not fate. Although the more refined level II procedure does include fate, it is not yet operational in the form of characterisation factors. This level also requires detailed information on the local situation (including the distance between emission source and 'receptor'). In most LCA studies this information will not be available, and the level II method therefore seems more suitable for local risk analysis than for LCA. The characterisation factors developed by Solberg-Johansen (1998) for the risks of radioactive solid waste are not fully described in her thesis. Consequently, they could not be properly assessed. Moreover, they

require information on the composition of the nuclear waste to be collected in the inventory, and this is often not available.
The baseline characterisation method for impacts of ionising radiation, the method of Frischknecht *et al.* (2000), is evaluated with respect to the (ISO-based) criteria adopted in this Guide in Table 4.3.15.1.

Table 4.3.15.1: Evaluation of the baseline characterisation method for impacts of ionising radiation, the method of Frischknecht *et al.* (2000), with respect to the (ISO-based) criteria of Table 4.3.1.

	criterion	evaluation
1	scientifically and technically valid	yes, the characterisation factors are based on common dispersion models and pathways of human exposure
2	environmentally relevant	yes, endpoint level
3	internationally accepted	no
4	value-choices and assumptions	present
5	focal point in environmental mechanism	endpoint
6	linearity	yes
7.	time span	100,000 years
8	fate, exposure and effects	all included
9	less is better	yes, effects below threshold are also included
10	time- and location-independent	representative for France
11.	operational	yes, for 49 radioniclides (including Radon–222)
12	uncertainty margins	unknown

The following method is included in the guide as an option for sensitivity analysis.
- If there are substantial emissions of radionuclides that are not included in Frischknecht *et al.* (2000), the level I procedure of Solberg-Johanson (1998) for emissions may provide valuable additional information, as it covers more nuclides. In such cases it is recommended to use this level I procedure for emissions.

Recomendations for extended LCAs
- The level II procedure of Solberg-Johansen (1998) can be used to augment the baseline method if radionuclide emissions prove to be important in the results of the Impact assessment step and detailed local information on these emissions is available.

method status	characterisation method/factor	reference
baseline	ionising radiation damage factors	Frischknecht *et al*., 2000
alternative	screening factors – level I	Solberg-Johansen, 1998
additional	screening factors – level II	Solberg-Johansen, 1998
variant	–	–

RESEARCH RECOMMENDATIONS
Long-term research:
- No method has yet been developed to assess direct external exposure to radiation, from building materials, for example. It is recommended to investigate the scope for integrating assessment of such exposure into the methods for exposure to releases of radioactive substances.
- Further research is necessary to assess whether current emissions of radionuclides have a significant impact on the natural environment. If this is the case, current methods for assessing the impacts of ionising radiation should be refined.
- Further research is also required to determine whether the method of Solberg-Johansen (1998) for assessing the risks of radioactive waste can be integrated with the method of Frischknecht *et al.* (2000). Moreover, due efforts should be made to obtain a more detailed picture of how both the short- and the long-term risks of both solid and liquid waste can be incorporated in the method.

4.3.16 Casualties

TOPIC

This impact category refers to casualties resulting from accidents. The area of protection is human health (see Figure 4.2.2). In Heijungs *et al.* (1992) this impact category was referred to as 'direct victims', but below this unfortunate term has been replaced by 'casualty'.

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)

In the classification step, various categories of casualty would have to be weighted if casualties other than fatalities (slightly injured, seriously injured and so on) were to be included in the Inventory analysis. At the time the 1992 guide was being prepared there were no methods for such weighting, nor indeed was such a method needed, as graded data on accident victims were lacking. For this reason the Heijungs *et al.* (1992) assessed and aggregated casualties by using a characterisation factor 1:

$$Casualties = C \qquad (4.3.16.1)$$

where C is the number of casualties (dimensionless) listed in the inventory table.

Most LCIA-methodologies do not include an impact category 'casualties'. Schmidt & Brunn Rasmussen (1999) describe a very useful method for including the working environment in LCA which encompasses casualties. It is based on a database developed by EDIP in which the working environment impacts per kilo of produced goods are listed for a number of economic activities.

PROSPECTS

If and when an impact category is distinguished aggregating all human health impacts, in the form of DALYs (see textbox in Section 4.2), casualties might be assigned to this new category. The method of Schmidt & Brunn Rasmussen (1999) might also be useful for including various impacts on human health in the working environment.

CONCLUSIONS

Although the method of Schmidt & Brunn Rasmussen (1999) seems a promising starting point for developing a characterisation method for casualties, the scope of the present project precluded an in-depth evaluation. We therefore recommend using the method of Heijungs *et al.* (1992), in which the number of fatal casualties is simply multiplied by a charaterisation factor of 1. In the future it is recommended to assign casualties to the impact category human health. The method of Heijungs *et al.* (1992) is evaluated with respect to the (ISO-based) criteria adopted in this Guide in Table 4.3.16.1.

Table 4.3.16.1: Evaluation of the baseline characterisation method for casualties, using characterisation factor = 1, with respect to the (ISO-based) criteria of Table 4.3.1.

	criterion	evaluation
1	scientifically and technically valid	not relevant
2	environmentally relevant	category indicator is close to intervention
3	internationally accepted	no
4	value-choices and assumptions	no
5	focal point in environmental mechanism	close to intervention
6	linearity	yes
7	time span	not relevant
8	fate, exposure and effects	not applicable
9	less is better	yes
10	time- and location-independent	yes
11	operational	yes
12	uncertainty margins	very low

No methods are included in this Guide as options for sensitivity analysis.

No recommendations are given for extended LCA.

method status	characterisation method/factor	reference
baseline	unweighted aggregation of victims	Heijungs *et al*., 1992
alternative	–	–
additional	–	–
variant	–	–

RESEARCH RECOMMENDATIONS

Short-term research:

- Research should be initiated to investigate the scope for including the method of Schmidt & Brunn Rasmussen (1999) for the working environment.

Long-term research:

- Research is required to develop a method to assign casualties to a future impact category 'human health'.

4.3.17 Interventions for which characterisation factors are lacking

TOPIC

Many practical cases will involve emissions of toxic chemicals for which no toxicity potentials are listed in the tables with characterisation factors. The same applies to acidifying substances, ionising substances, depletable resources and so on. A general guideline given for extended LCAs is to calculate, estimate or extrapolate missing characterisation factors. This will often be unfeasible, however, for lack of time or knowledge, for instance. In such cases these overlooked items should be discussed in a separate part of the impact assessment

A distinction should be made between:

- interventions known to contribute to an impact category and for which no characterisation factor is available but for which a factor can be calculated, estimated or extrapolated;
- interventions known to contribute to an impact category but for which no characterisation factor can be found, calculated, estimated or extrapolated;
- interventions assumed to be environmentally relevant but not contributing to any of the selected impact categories;
- interventions assumed not to be environmentally relevant.

As far as possible, all interventions for which no characterisation factors are available should be assigned to all impact categories to which they are known to contribute, based on the best available knowledge. These interventions shall than be listed in the environmental profile under the appropriate impact category. If possible, a qualitative or quantitative estimate of their potential environmental impact should be given. Quantitative estimation can, for example, be accomplished by finding an intervention similar in terms of chemical structure for which (a) characterisation factor(s) is/are available.

If it is not known to which impact category an intervention should be assigned it shall be listed in a separate category 'Interventions for which characterisation factors are lacking'.

Interventions anticipated to be environmentally irrelevant may be excluded from the environmental profile, but this should be transparently justified in the LCA study report.

CONCLUSIONS

Interventions for which characterisation factors are lacking should be dealt with in the following manner:

- Interventions for which a characterisation factor can be calculated, estimated or extrapolated should be included in the environmental profile under the relevant impact category, accompanied by a clear explanation of the divergent status of the characterisation factor and the method used to obtain it.
- Interventions for which no characterisation factor can be calculated, estimated or extrapolated but which are known to contribute to one or more impact categories should be included in a separate part of the environmental profile labeled 'Interventions for which characterisation factors are lacking', accompanied by all relevant additional information such as:
  - substance name;

- emission compartment;
- amount emitted;
- impact category to which a contribution is suspected;
- if possible, an indication of the significance of the suspected impact.

– Interventions known to be of environmental relevance but contributing to an impact category that is not selected should be included in the environmental profile in the same way.

Interventions expected to be environmentally irrelevant can be excluded from the environmental profile, but this should be transparently justified in the LCA study report.

4.3.18 Economic flows not followed to system boundary

TOPIC

LCAs may comprise certain flows that are not specified in terms of environmental interventions, either inputs, like energy or materials, or outputs, like solid waste. Every effort should be made to avoid such flows, in the first place by applying the data estimation methods outlined in Section 3.8. All economic flows that cannot be followed to the system boundary should then be listed in a separate category: 'Economic flows not followed to the system boundary'. Flows listed in this category should always be described qualitatively (e.g. 'hazardous waste' and 'non-hazardous waste') and, wherever possible, quantitatively (e.g. 10^{-12} truck).

CONCLUSIONS

All economic flows that cannot be followed to the system boundary, even after application of the data estimation methods outlined in Section 3.8, should be listed in a separate category 'Economic flows not followed to the system boundary'.

RESEARCH RECOMMENDATIONS

No research foreseen.

4.4 Classification

TOPIC

In this step the environmental interventions qualified and quantified in the Inventory analysis are assigned on a purely qualitative basis to the various pre-selected impact categories (see Section 4.2). For a baseline list of interventions, for which characterisation factors have previously been derived, the classification step involves no actual work as these interventions have already been assigned to the various impact categories in this Guide (see Section 4.4 of Part 2b of this Guide). In the case of other interventions the practitioner will have to adopt an appropriate procedure of his own.

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)

Classification was not a separate step in Heijungs *et al.* (1992). At that time classification was the name given to the step covering the current ISO steps "Selection of impact categories, category indicators and models", "Classification" and "Characterisation". In Heijungs *et al.* (1992) interventions were assigned implicitly (i.e. not in a separate step) to relevant impact categories by means of associated characterisation factors. Interventions with multiple impacts, i.e. contributing to more than one impact category (parallel or serial), were assigned in their entirety to all relevant impact categories.

With respect to the classification step ISO 14042 (2000E) states: "When LCI results are assigned to impact categories, issues associated with LCI results may be highlighted. Assignment of LCI results to impact categories should consider the following, unless otherwise required by the goal and scope:

– assignment of LCI results which are exclusive to one impact category;
– identification of LCI results which relate to more than one impact category;

- distinction between parallel mechanisms, e.g. SO_2, is allocated between the impact categories of human health and acidification;
- allocation among serial mechanisms, e.g. NO_x may be assigned to ground level ozone formation and acidification.

If LCI results are unavailable or of insufficient data quality for the LCIA to achieve the goal and scope of the study either an iterative data collection or an adjustment of the goal and scope is required."

Guinée (1995), Lindfors *et al.* (1995c), Udo de Haes *ed.* (1996) and Wenzel *et al.* (1997) also discuss the topic of multiple impacts of chemical releases and together distinguish the following four categories of emissions:

- Emissions with parallel impacts, i.e. emissions of substances that may theoretically contribute to more than one impact category but in practice only to one, e.g. an emission of SO_2 which may have either toxic or acidifying impacts.
- Emissions with serial impacts, i.e. emissions of substances that may in practice have successive impacts, e.g. emissions of heavy metals which may first have ecotoxicological impacts and subsequently, via food chains, impacts on human health.
- Emissions with indirect impacts, i.e. emissions of substances having a primary impact that in turn leads to one or more secondary impacts, e.g. aluminium toxicity induced by acidification, or methane contributing to photo-oxidant formation, with the produced ozone contributing in turn to climate change, which in turn may contribute to stratospheric ozone depletion.
- Emissions with combined impacts, i.e. emissions of substances having a mutual influence on each other's impacts, e.g. synergistic or antagonistic impacts of toxic substance mixes, or NO_x and VOC, both of which are required for photo-oxidant formation.

In order to avoid double counting, for emissions having parallel impacts it is generally recommended in the literature that the respective contributions of such emissions to relevant impact categories be specified. However, no guidelines are available on how this task is to be performed.In general, such specification should be performed only in those cases where it really matters (where the contribution of the substance to one impact category substantially lessens its potential contribution to another, e.g. acidification or eutrophication by NH_3. Rough calculations show that SO_2, for example, is less relevant in this respect; see Heijungs *et al.*, 1992). If it is unclear how such emissions are to be allocated, as a general recommendation it is advised either to assign them in their entirety to *all* relevant impact categories or to divide them equally across the categories (e.g. 50/50). If there were one, all-encompassing fate and exposure model available covering all impact categories, rather than the diversity of models used for the various impact categories today, parallel impacts would no longer constitute a problem. Current fate and exposure models specify the compartment (or target organism) in which the substance has its principal impact and which impact categories are thus potentially relevant (e.g. emissions to the air may end up in soil or water, with consequent terrestrial or aquatic ecotoxic impacts, respectively). Given that we are still some way from having an all-encompassing fate and exposure model, however, full assignment of emissions having parallel impacts may, for the present, lead to some measure of double-counting.

For emissions having serial and indirect impacts the literature generally recommends allocating such emissions in their entirety to all relevant (i.e. serial and indirect) impact categories unless characterisation factors for this purpose are lacking, as in the case of missing (indirect) GWP factors, for example.

For emissions having combined impacts the literature generally recommends introducing assumptions regarding background concentrations of the other relevant substances. In practice this is currently only feasible for NOx as a precursor in photo-oxidant formation, but not for synergistic or antagonistic impacts of toxic substance mixes, as knowledge on these issues is virtually entirely lacking.

Section 4.4 of Volume 2b lists all interventions for which at least one baseline characterisation factor is available. We advise practitioners to use this list as a default and classify inventory results according to the choices embodied in it. Items not included in the default list should be included using expert judgement and other literature sources.

PROSPECTS

As indicated above, the subject of parallel impacts may eventually become part an all-encompassing fate and exposure model.] In that case the 'Classification' step will be restricted solely to the assignment of interventions to defined impact categories in Section 4.2 (similar to Characterisation now).

CONCLUSIONS

Emissions having truly parallel impacts are probably rather scarce. As an all-encompassing fate and exposure model is still lacking, it is recommended to assign such emissions in their entirety to all relevant impact categories.

Emissions with serial and indirect impacts should also be assigned in their entirety to all relevant impact categories, unless there is insufficient information to do so (e.g. in the case of missing (indirect) GWP-factors) or an overlap between impact categories (e.g. aluminium toxicity induced by acidification, which is already regarded as part of the impact category acidification).

Emissions with combined impacts should likewise be assigned in their entirety to all relevant impact categories. In characterisation modeling assumptions must then be made regarding standard concentrations of other relevant substances. Synergistic and antagonistic impacts of toxic substance mixes cannot currently be covered in LCIA.

RESEARCH RECOMMENDATIONS

Short-term research

- It is recommended to undertake research to find appropriate default values for the division of parallel interventions, where relevant.

Long-term research

- Research on procedures for assigning new substances to impact categories is recommended.

4.5 Characterisation

TOPIC

In the characterisation step of Impact assessment the environmental interventions assigned qualitatively to a particular impact category in classification are quantified in terms of a common unit for that category, allowing aggregation into a single score: the indicator result.

DEVELOPMENTS IN THE LAST DECADE

According to ISO 14042 (2000E) this step is now concerned only with calculation of the (category) indicator results, using the methods described in Section 4.3. For each impact category the (category) indicator result is calculated by multiplying the relevant interventions by their corresponding characterisation factors. Together, these results constitute the 'environmental profile': a table showing the indicator results for all the predefined impact categories supplemented by any other relevant information.

Heijungs *et al.* (1992)

In Heijungs *et al.* (1992) indicator results were also calculated using characterisation factors:

$$indicator\ result_{cat} = \sum_i m_i \times characterisation\ factor_{cat,i} \qquad (4.5.1)$$

where i is the type of intervention (e.g. substance emission or resource extraction) and m_i its magnitude.

PROSPECTS

No further developments are foreseen.

CONCLUSIONS

For each impact category the (category) indicator results are calculated by multiplying the relevant interventions by their corresponding characterisation factors, according to the formulae elaborated in Sections 4.3.1 to 4.3.16.

RESEARCH RECOMMENDATIONS
No research is foreseen.

4.6 Normalisation

TOPIC
ISO 14042 (2000E) defines normalisation as "calculation of the magnitude of indicator results relative to reference information". The reference information may relate to a given community (e.g. The Netherlands, Europe or the world), person (e.g. a Danish citizen) or other system, over a given period of time. Other reference information may also be adopted, of course, such as a future target situation. While normalisation is considered an optional element of LCIA in ISO 14042 (2000E), in this Guide it constitutes a *recommended* step of life cycle Impact Assessment

The main aim of normalising the (category) indicator results is to better understand the relative importance and magnitude of these results for each product system under study. Normalisation can also be used to check for inconsistencies, to provide and communicate information on the relative significance of the (category) indicator results and to prepare for additional procedures such as weighting or Interpretation (ISO 14042, 2000E).

There exist other definitions of 'normalisation', as is the case in multicriteria analysis (where it if often understood as division of the various values in a data set by a single reference value from that set, to express all the values in terms of that reference value). These definitions and the associated methods are not discussed here.

DEVELOPMENTS IN THE LAST DECADE
ISO 14042 (2000E) states that in selecting the reference system due consideration should be given to

> Heijungs *et al.* (1992)
> In Heijungs *et al.* (1992) the normalisation step was mentioned and defined, but not yet numerically elaborated.

the consistency of the spatial and temporal scales of the environmental mechanisms and of the reference value. It is stated, furthermore, that it may be desirable to use several reference systems, in order to assess their respective influence on the LCIA outcome, and that the normalised results are to be termed 'the normalised LCIA profile'. (For reasons of consistency in this Guide the term 'normalised environmental profile'[1] is employed; see Glossary).

The reference value is the indicator result for a reference system. It is thus for a given impact category the sum of all the interventions associated with the reference system multiplied by the appropriate characterisation factors:

$$\textit{indicator result}_{cat,ref} = \sum_i m_{i,ref} \times \textit{characterisation factor}_{i,cat} \qquad (4.3.18.1)$$

$$\textit{normalised indicator result}_{cat} = \frac{\textit{indicator result}_{cat}}{\textit{indicator result}_{cat,ref}} \qquad (4.3.18.2)$$

with:

$\text{indicator result}_{cat,ref}$ the indicator result for impact category *cat* and reference system *ref* (e.g. in $kg \cdot yr^{-1}$ or in $kg \cdot yr^{-1} \cdot capita^{-1}$); the reciprocal of indicator $\text{result}_{cat,ref}$ is here referred to as the normalisation factor for impact category *cat* and reference system *ref*;

$m_{i,ref}$ the magnitude of intervention *i* (emission, resource extraction or land use) associated with the reference system *ref* (e.g. in $kg \cdot yr^{-1}$ or in $kg \cdot yr^{-1} \cdot capita^{-1}$);

[1] This was necessary because 'LCIA' refers to the entire Impact assessment phase, which comprises a number of quite distinct steps, such as classification, characterisation, normalisation and weighting, each withits own results.

characterisation factor$_{i,cat}$ the characterisation factor for intervention *i* and impact category *cat* (e.g. in kg·kg^{-1});
normalised indicator result$_{cat}$ the normalised indicator result for impact category *cat* (in yr or in yrcapita);
indicator result$_{cat}$ the indicator result for impact category *cat* (e.g. in kg).

There is currently much discussion regarding appropriate choice of reference systems for normalisation. ISO provides several examples of the type of reference system that can be used for the purposes of normalisation (ISO 14042, 2000E);

1. the aggregate interventions for a given area (global, regional, national or local) in a reference year; examples are the normalisation figures of Guinée (1993) for the world and those of Blonk *et al.* (1997b) for the Netherlands;
2. the per capita interventions for a given area in a reference year; examples are the per capita figures for Danish and world citizens of Wenzel *et al.* (1997);
3. a baseline scenario, such as the calculated (category) indicator result for a given alternative product system; no examples of this method are available.

Blonk *et al.* (1997b) have developed a fourth type of reference value:

4. the aggregate interventions associated with the habits of consumption of a particular population in a reference year, for example the data for the Dutch population of Blonk *et al.* (1997b).

The first two approaches are most commonly used. Which is preferable depends on another important choice: normalisation at a single scale level (generally the world, sometimes Europe) versus combined normalisation at the global and regional scales.

Normalisation at a single scale level

In this option the results for each impact category are normalised against reference data from the same reference area, either aggregate interventions (method 1) or per capita interventions (method 2). The most suitable reference area would then seem to be 'the world', as LCAs are not generally site-dependent. The two methods differ only by a constant factor (*1 / world population*) and which method is adopted will depend on the goal of the study. Reference against aggregate world interventions has the advantage of providing a comparison with the actual aggregate magnitude of the impact categories at stake, while comparison with the interventions of an 'average world citizen' shifts the focus of issues to the individual and may therefore be better for communicative purposes. Normalisation data for the world in 1990, Europe in 1995 and the Netherlands in 1997 are available in Huijbregts *et al.* (in prep.) The EDIP project is currently developing estimates for the contribution of an 'average world citizen' to regional and local impacts as reference values (Hoffman & Stranddorf, 1999). At the time of writing of this Guide these data were not yet available, however.

Normalisation at different scale levels

If, in a particular LCA study, Inventory analysis and Impact assessment for global impact categories are carried out on a global scale and Inventory analysis and Impact assessment for regional impact categories on a regional scale, there are two options for normalisation:

- the results for each impact category are normalised using reference data from the same reference area (see above); or
- normalisation is carried out at two scale levels, with the results for global categories being normalised using global reference values and the results for regional categories using the appropriate regional reference values. This approach is more in line with ISO 14042 (2000E). ISO states: "The selection of the reference system should consider the consistency of the spatial and temporal scales of the environmental mechanism and the reference value".

In this second case, simple division by the aggregate interventions per area would not be correct, for the results for global impact categories would then be rated far lower than those for regional impact categories, because aggregate world interventions far outstrip interventions from any one region. The per capita approach (method 2) resolves this problem. The world totals are divided by the total world population and the regional totals by the population of the region in question.

An example of an LCA method in which global and regional Impact assessment, normalisation and weighting are combined is the EDIP approach (Wenzel *et al.*, 1997). In EDIP, global impact categories are assessed using global characterisation factors, normalisation is based on global per capita figures, and weighting is based on a distance-to-target approach using global reduction targets. Regional impact categories are assessed using Danish characterisation factors; normalisation is based on Danish per

capita figures and weighting (see Section 4.8) is based on a distance-to-target approach using Danish reduction targets.

It is stressed once more that if regional normalisation is performed using different reference areas, the results for regional impact categories in regions with relatively high per capita interventions will be rated lower than the same results for regions with relatively low per capita interventions. In such cases, then, not only normalisation but also weighting or grouping (see Section 4.7) of the normalised results should be performed on the same regional basis (Heijungs, 1997b). In this kind a regional set-up, inventory data as well as characterisation, normalisation and weighting factors are therefore required for each and every region involved in the study. Regional characterisation factors for acidification and terrestrial eutrophication are presently available (and POCPs for high and low NO_x background concentrations). As yet, however, there is no uniform set of regional reference values that can be applied for these impact categories. For the regions Denmark and the Netherlands these can probably be readily developed based on regional intervention data collected by Wenzel *et al.* (1997) and Blonk *et al.* (1997b). For other regions, however, no reference values are available. Regionalised weighting factors are even scarcer.
If grouping or weighting are performed on a different regional basis than that used for normalisation, interpretation of the results of the LCA study will become problematical.

Temporal horizon
Another point to be considered is the ISO statement that reference system adopted "should consider" not only consistency in terms of spatial scale, but also with respect to "the temporal scales of the environmental mechanisms and reference value". This is not specified in any further detail. However, it seems undesirable to relate results for impact categories with shorter-term effects, such as photo-oxidant formation, to reference systems for shorter periods. e.g. 5 days, and at the same time relate those for impact categories with long-term effects, such as climate change, to reference systems based on longer periods, e.g. 100 years (Heijungs, 1997b). The temporal horizon of the reference system is not the reference point for the *impacts* occurring in a particular period but for the *interventions* in that period. As yet, moreover, all available reference values are based on a single period (generally one year).

Consistency
Last but not least, there is the issue of consistency between the methodological choices made in calculation of normalisation factors and those made in the LCA study in which the normalisation factors are being applied. In principle, the same characterisation factors should be used; system boundaries should be treated similarly, e.g. with respect to dredging[1], agriculture and landfills (see Section 3.2); cut-off (see Section 3.8); multifunctionality and allocation (see Section 3.9); etc. In practice, however, it is often unclear how such choices have effectively been made in the emission figures available for use as a basis for normalisation. The practitioner should at any rate be aware of possible inconsistencies and their potentially major influence on the significance of results and do all he or she can to avoid such inconsistencies wherever possible. If necessary, the normalised environmental profile should be accompanied by appropriate comments.

PROSPECTS
Besides the various purposes outlined above, normalisation may also serve another function: to 'compensate' for interventions on which data are lacking. In the Inventory analysis phase of an LCA it may be convenient for an interested party to 'forget' certain emissions or other interventions. This problem can then be tackled by using a similarly incomplete subset of interventions as a reference for normalisation. If different subsets of interventions are considered for each of the processes examined in an LCA, this would mean performing normalisation for each process individually, prior to aggregation of the data in a single inventory table. This would probably require additional efforts. The subset used for normalisation should then be the same as that employed for the Inventory analysis of the processes in question. In other words, interventions included in the Inventory analysis but with a zero result should be included in the normalisation subset, while interventions ignored in the Inventory analysis should also be

[1] Including dredging as an economic activity (see Section 3.2) implies that the removal of sediments should be regarded as a negative (sediment) emission and the dumping of sediments as a positive emission to soil. The net result is then a shift between compartments of a given amount of substance(s). For practical reasons this shift is now often ignored in the normalisation step.

excluded. Whether or not such partial normalisation is a useful option should then be examined in further detail.
On a second topic, a number of countries are developing (regional) normalisation data sets which will become available in due course.

CONCLUSIONS

We recommend using normalisation data based on one geographically and temporally well defined reference system, preferably the world for one year (consistent with the temporal coverage of the Goal and scope definition phase of the study), for all impact categories. The method based on aggregate world interventions (method 1) or that based on the interventions of an 'average world citizen' (method 2) are both applicable. Which one is chosen depends on the goal of the LCA study and therefore both values are given in this Guide.
In Volume 2b Section 4.6, normalisation data for the world in 1990, Europe in 1995 and the Netherlands in 1997 are presented for use as normalisation factors for each impact category for each of the baseline characterisation methods recommended in this Guide (see Section 4.3). These data are based on Huijbregts *et al.* (in prep.). If other characterisation methods are used the unaggregated data (interventions per reference area and time) can be used to calculate the appropriate normalisation factors for these methods according to the formulae given above. In principle, normalisation might alternatively be carried out at two scale levels, with results for global impact categories being normalised on the basis of global reference values and the results for regional impact categories on the basis of appropriate regional reference values. However, a uniform set of regionally specified reference values is still lacking, as observed above. If different scales are combined:

- only per capita normalisation data should be used;
- normalisation data for regional impact categories should be based on the regions where the interventions of the LCA study at stake took place;
- if grouping or weighting is performed, the regionally normalised data should be grouped or weighted using regional grouping methods or regional weighting factors.

Finally, the practitioner should be aware of possible inconsistencies between the methodological choices (with respect to system boundaries and so on) underlying the calculation of normalisation factors and the methodological choices made in the LCA study in which the normalisation factors are being applied. Inconsistencies should be avoided as far as possible. For example, if the practitioner wishes to apply non-baseline characterisation factors, the corresponding set of non-baseline normalisation factors should be applied from the spreadsheet mentioned in Section 4.6 of part 2a of this Guide.

RESEARCH RECOMMENDATIONS

Short term research:

- A study to collect adequate world data, based as far as possible on empirical measurements and derived statistics, is highly recommended.
- The pros and cons of partial normalisation, i.e. normalisation factors based on a subset of interventions, in the case of incomplete datasets (unit processes for which interventions are missing) should be further investigated. These subsets need not necessarily be the same for each unit process.

4.7 Grouping

TOPIC

Grouping is a step of Impact assessment in which impact categories are aggregated into one or more sets. It is an optional element for which two possible procedures are available: sorting and ranking, defined by ISO as follows (ISO, 14042, 2000E):

- sorting of the category indicators on a nominal basis e.g. by characteristics such as emissions and resources or global regional and local spatial scales;
- ranking of the category indicators on an ordinal scale, e.g. a given order or hierarchy, such as high, medium and low priority (ranking is based on value-choices).

It should be noted that this ISO definition of grouping relates to valuation of the *impact categories*, not the indicator results. Indicator results can also be ranked without placing any value on the impact category as such. The results are then ranked purely on the basis of the magnitude of the result itself (5 is higher than 3). For this type of ranking, the methods developed in the area of multi-criteria analysis may be useful (see also Section 4.8).
ISO states, furthermore, that application and use of grouping methods "shall be consistent with the goal and scope of the LCA study and it shall be fully transparent". As different individuals, organisations and societies may have different values, it is well feasible that different parties will arrive at different ranking results based on the same indicator results. ISO provides no further examples of grouping and ranking methods.

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)
In Heijungs *et al.* (1992) grouping in the sense of the ISO definition was not mentioned.

Sorting

No further developments on the subject of sorting are known to the authors of this Guide.

Ranking

ISO 14042 (2000E) explicitly mentions the fact that ranking is based on value-choices.

In a paper on weighting methods, Finnveden (1999a) identified and reviewed available methods for grouping impact categories in LCA. A variety of criteria are employed for this purpose, summarised by Schmitz *et al.* (1994) as follows:
- 'ecological threat potential;
- reversibility - irreversibility;
- global, regional, local;
- environmental preference of the population;
- relationship of actual and/or previous pollution to quality goals.'

Based on these criteria these authors group impact categories into five classes, labeled as being from 'lesser importance' to 'very large importance'. Volkwein *et al.* (1996) suggest that impact categories be qualitatively ranked by an expert panel, employing three criteria: scale of impact (anticipated area affected), reversibility time (time required for restoration of damage after the intervention ceases) and actual hazard the (potential) impact (in terms of assessments of actual exceedance of standards, uncertainty about the extent of harm, etc; see Volkwein *et al.,* 1996).

Finnveden (1999a) warns that in many cases the results of grouping (and especially ranking) will not be reproducible and that the individual preferences of those performing such an exercise will have a major influence on the overall results of the LCA. It can be argued, moreover, that grouping will generally be useful only in the context of reaching a final decision in a specific LCA study. In such cases, however, there will still be a subsequent need for some kind of weighting, as grouping does not yield a single score.

PROSPECTS

Based on the recent ISO developments it is to be expected that grouping and ranking methods will be further developed and that experience will be gained with application thereof.

CONCLUSIONS

Grouping is an optional element of LCA according to ISO 14042 (2000E). Grouping is likely to be highly influenced on the personal preferences of practitioners, and little work has yet been done on operationalisation its usefulness is in doubt. Therefore no specific method is recommended in this Guide.
No methods have yet been published for 'sorting', For 'ranking' impact categories, the five criteria used by Schmitz *et al.* (1994) can serve as a starting point. The criteria used to evaluate impact categories should be described extensively. It is recommended to present the results of grouping as a matrix on the basis of the criteria employed.

RESEARCH RECOMMENDATIONS

Short-term research

- In the short term a review should be prepared of grouping and ranking methods used in other tools or in the social sciences that may be useful in the LCA context, too. As the details of whether and how to differentiate between impact subcategories and categories in grouping are as yet unclear, this should also be analysed.

Long-term research

- Some of the methods found in the research mentioned above may be subsequently elaborated for practical application in LCA.

4.8 Weighting

TOPIC

Weighting is an optional step of Impact assessment in which the (normalised) indicator results for each impact category assessed are assigned numerical factors according to their relative importance, multiplied by these factors and possibly aggregated. Weighting is based on value-choices (e.g. monetary values, standards, expert panel). A convenient name for the result of the weighting step is 'weighting result', of which there is generally one for each alternative product system analysed. As a variation, though, weighting may also yield several weighting results per product system, for instance for human health, ecosystem health and resources. The term 'weighting profile' is used in this Guide for the overall result of the weighting step: a table showing all the weighting results, supplemented by any other relevant information.

Before weighting can be performed, the various indicator results must first be converted into the same units, one possible method for which is normalisation (see Section 4.6).

DEVELOPMENTS IN THE LAST DECADE

Heijungs *et al.* (1992)

In Heijungs *et al.* (1992) two weighting methods were discussed: quantitative and qualitative multi-criteria analysis (MCA), but were not elaborated. Rather, their respective advantages and disadvantages were compared with those associated with unweighted comparison.

Table 4.8.1: Evaluation of different weighting methods.

	unweighted comparison	qualitative MCA	quantitative MCA
convincing	-	+	+
includes qualitative aspects	+	+	-
reproducible	+	-	+
less open to discussion	+	-	#

+: yes, -: no, #: moderate.

Heijungs *et al* (1992) state that quantitative MCA is preferable as it provides greater transparency. At the time of writing, this method was used only to a limited extent, if at all, however. Hence, it was recommended to dedicate a separate section in each LCA report to a discussion of the reasons for preferring one product alternative over another.

ISO 14042 (2000E) explicitly mentions the fact that weighting is based on value-choices and not on the natural sciences. Under the heading 'weighting', ISO again states that "the application and use of weighting methods shall be consistent with the goal and scope of the LCA study and shall be fully transparent". As different individuals, organisations and societies may have different values, it is possible that different parties will arrive at different weighting results based on the same indicator results.

ISO also states that "all weighting methods and operations used shall be documented to provide transparency". Inventory results and the (normalised) environmental profile arrived at prior to weighting

are also to be made available, together with the weighting results. This ensures that (ISO 14042, 2000E):

– "trade-offs and other information remain available to decision-makers and to others; and
– users can appreciate the full extent and ramifications of the results."

Finally, ISO states that weighting shall not be used for comparative assertions disclosed to the public. No examples of weighting are provided by ISO. Outside ISO, however, weighting methods have received extensive attention since 1992.

In addition, qualitative information, non-weighted indicator results and non-characterised interventions should be also made available as part of the weighting profile (see above; addition by authors of this Guide).

Finnveden (1999a) has published a review of weighting methods used in LCA. Since this is the most recent review article, it is summarised here fairly extensively (see textbox below).

Finnveden (1999) classifies weighting methods into five main groups, based on Lindeijer (1996) and Braunschweig (1996). Three of these are relevant for the weighting of impact categories in LCA:

1. methods based on monetary values;
2. methods based on (government) standards or targets;
3. methods based on the judgment of an authoritative panel.

1. Methods based on monetary values

There exist a wide variety of approaches based on monetary value, which Finnveden classifies as follows:

a) Willingness-to-pay methods:
   Methods based on the willingness to pay a certain amount of money to avoid the occurrence of an intervention, threat or damage, with three basic variations:
   - Individual revealed preferences
     These include approaches based on the travel costs incurred in going to recreational areas, the price paid for houses in a given area, and variations in wages depending on the risks associated with different types of jobs. This is essentially valuation on the basis of market values.
   - Individual expressed preferences
     This approach is based on asking individuals to explicitly attach a value to environmental assets ('contingent valuation method').
   - Collective revealed preferences
     These can be derived from political and government decisions. Society's willingness-to-pay can for example be derived from the expenditure devoted to saving a 'statistical life', from the costs incurred to reduce an emission to a set emission limit, or from 'green taxes'.

b) Other monetary methods, i.e. not based on willingness-to-pay
   These methods are based on estimates of cost, but with no indication that any individual might be willing to pay that cost. Examples are the cost of reducing emissions to a future target level or the cost of restoring damages.

In the examples given by Finnveden the difference between 'collective revealed preferences' and 'other monetary methods' is rather vague.

Since the various kinds of monetarisation methods are concerned with differing kinds of values, the respective methods yield different results and a sum expressed in monetary units cannot simply be compared directly with another sum in the same units. For example, the total economic value measured by the contingent valuation method is typically higher than the value derived from market valuations.

Finnveden (1999) describes three monetary methods in more detail: EPS (Steen, 1996), Tellus (1992) and ExternE (EC, 1995a; Van Beukering *et al.*, 1998). Although a weighting method based on monetary valuation is in itself promising, none of these methods could be recommended for LCA (Finnveden, 1999a). The EPS method is a hybrid of the outlined monetary approaches and thus combines different values (market prices and other values). As stated, this is not to be recommended, however. The method also has large data gaps, suffers from lack of transparency (e.g. valuation of biodiversity) and contains some scientific, logical and computational errors. Moreover, the EPS method yields results that do not reflect the common perceptions of society. The Tellus method is based on a collective revealed preference approach. The method has several data gaps and elements in need of updating and cannot be used for the impact categories employed in this Guide. The ExternE method is based mainly on contingent valuation methods (i.e. individual expressed preferences), although elements of other valuation methods are also used. The method still has some data gaps and parts in need of updating. A further problem associated with methods based on the modeling of damage, such as ExternE (but also Eco-Indicator '99; see below), is the major risk of underestimation because of unknown damage pathways (see textbox in Section 4.2).

Finnveden (1999) mentions several other monetary methods (e.g. Krozer, 1992; Huppes *et al.*, 1997). However, insufficient details were published to evaluate these properly. Another interesting approach is provided by Beetstra (1998); within the scope of the present Guide it was no no longer possible to properly evaluate this method.

Finnveden recommends no one weighting method, observing that many methods suffer from serious data gaps. However, he considers the panel and monetary approaches the most promising.

Heijungs (1994a) has noted that if weighting is based on an authoritative panel, the panel must be provided with carefully prepared information and questions, results being highly influenced by the manner in which questions are formulated. At the least, the panel must have adequate qualitative and/or quantitative information about the relationship between the chosen category indicators and endpoints (i.e. the environmental relevance of the category indicators).

2. Methods based on (government) standards or targets
These methods are often referred to as 'distance-to-target methods' and differ in the equation used to relate target to weighting factor, the target itself, and the choice to weight inventory results or characterisation results .
Finnveden (1999) describes a number of distance-to-target methods, viz. Ahbe *et al.*, 1990 (updated in BUWAL, 1998); Schaltegger & Sturm, 1991; Baumann, 1992; Corten *et al.*, 1994; Goedkoop, 1995; Kalisvaart & Remmerswaal, 1994; Kortman *et al.,* 1994; Wenzel *et al.*, 1997. Finnveden does not recommend using any of these distance-to-target methods because no inter-effect (inter-category) weighting is performed. The relative importance of the actual effects compared is not defined, or implicitly set to unity for each target. For this reason 'distance-to-target methods' are not in fact true weighting methods and their use is therefore often advised against

3. Methods based on the judgment of an authoritative panel
In these methods a group of people are asked to provide weighting factors on the basis of their specialist knowledge. This may be by way of a questionnaire, interviews or group discussions among a group of panellists: experts, stakeholders or laymen. A single- or multi-round procedure may be adopted, with or without feedback.
Finnveden (1999) describes a number of panel methods (including the Eco-Indicator '99 method), viz. Annema, 1992; Kortman *et al.*, 1994; Wilson & Jones, 1994; Nagata *et al.*, 1995; Puolamaa *et al.*, 1996; Lindeijer, 1997; Huppes *et al.*, 1997; Goedkoop & Spriensma, 1999. He does not recommend using any of these panel methods because they have generally been developed for specific cases and it is by no means clear whether they can be universally applied. Moreover, there are significant data gaps in many of these methods and most assign more or less equal weight to all categories.

Lindeijer (1996) poses the question whether weighting should be performed on a case-by-case basis using discourse procedures to involve stakeholders, or be based on a universal set of weighting factors. Lindeijer (1996) has no preference for any one weighting method, but distinguishes five criteria of importance for weighting indicator results, as follows:

- 'natural science information on the relationship between total burden ('actual flows')[1] and the extent of actual damages to objects[2];
- foreseeable future trends in flows;
- reversibility of the damage (including the time for spontaneous reversibility);
- importance of the damaged object (including scale of effect and substitutability);
- uncertainty on the extent of damage.'

Lindfors *et al.* (1995c) discuss and 'test' a number of weighting methods. Based on their findings they recommend using not one but several weighting methods. In Finnveden (1997) the author states that consensus will never be achieved on a universal weighting method because there is no consensus in society regarding the fundamental values influencing the choice of valuation method. He therefore recommends developing several methods simultaneously (Finnveden, 1997).

An entirely different approach is that of Lundie (1999), who evaluates product alternatives based on ranges/distributions of preferences obtained in surveys. These are applied to the indicator results instead of fixed weighting factors, with upper- and lower-bound evaluation being employed to draw conclusions.

[1] In the terminology of this Guide, this is the indicator result for a particular reference area and reference time (see Section 4.6).
[2] In the terminology of this Guide, objects correspond to endpoints (see Figure 4.2.2).

PROSPECTS

There is growing interest in using methods adopted from multi-criteria analysis for life cycle Impact assessment. 'Classical' weighting compares alternative product or system alternatives based on a single set of average weighting factors that are used as multipliers for the respective indicator results. Multi-criteria analysis focuses more on the ranking of product attributes (i.e. indicator results) and draws conclusions based on these results without weighting.

CONCLUSIONS

None of the monetary methods described by Finnveden can be applied to the impact categories elaborated in this Guide. In the EPS method and the Tellus method 'effect assessment' and valuation are interwoven, resulting in a factor per intervention rather than per impact category. In the ExternE approach, the weighting factors are based on characterisation methods very different from the ones defined in this Guide. The collective revealed preferences method presented by Huppes *et al.* (1997) covers only six impact categories and cannot therefore be used for all categories; the frequently important categories ecotoxicity and abiotic depletion are not included.

Although there are several distance-to-target methods that cover some of the relevant impact categories, none is applicable to all. These methods are based on differing national and international policy targets and are not readily combined. They do have one thing in common, however: the relatively heavy weight attached to the category of stratospheric ozone depletion. Most of these methods need to be updated to recent standards.

Similarly, although certain panel methods cover some of the relevant impact categories, none can be used for all. In the Eco-Indicator '99, for example, the weighting factors are based on characterisation methods different from those defined in this Guide and cannot therefore be applied here. Moreover, these panel methods are generally based on panels of stakeholders assembled for a specific case study.

In summary, we conclude that there is no complete and authorised weighting set available at the present time that can be used for the weighting of the impact categories elaborated in this Guide.

Bearing in mind the above, the following recommendations are made.

- Avoid weighting wherever possible. If the questions formulated in the Goal and scope definition can be adequately answered without weighting, for instance if one product alternative is clearly stands out above all others (e.g. because it scores better on all impact categories) weighting will be unnecessary. Note that according to ISO weighting shall not be used at all for comparative assertions disclosed to the public.
- If weighting is performed this should be done using a nationally or internationally authorised set of weighting factors covering all relevant impact categories. However, such a set is not currently available. We strongly recommend that a weighting set be developed covering all impact categories and approved by a panel having due international or national authority.

RESEARCH RECOMMENDATIONS

Short-term research

- In the short term a preliminary set of weighting factors should be developed that can be used for simplified LCA studies.
- The details of whether and how to differentiate between sub-impact categories and impact categories in weighting are still unclear and should be further examined.

Long-term research

- It should be investigated which methods (individual revealed preferences, individual expressed preferences, collective revealed preferences, expert panel, etc.) are most suitable for deriving a more definite nationally or internationally authorised set of weighting factors covering all relevant impact categories.

5. Interpretation

5.1 General introduction

In an LCA study the Goal and scope definition provides the initial groundplan of the study. The Inventory analysis supplies the data on relevant processes and interventions on which the assessment is to be based. In the Impact assessment phase the interventions are translated into potential environmental impacts. The final phase of an LCA is Interpretation (see). According to ISO 14040 in life cycle Interpretation "the results of a life cycle Inventory analysis and - if conducted - of a life cycle Impact assessment (LCIA) are summarised and discussed as a basis for conclusions, recommendations and decision making in accordance with the Goal and scope definition".

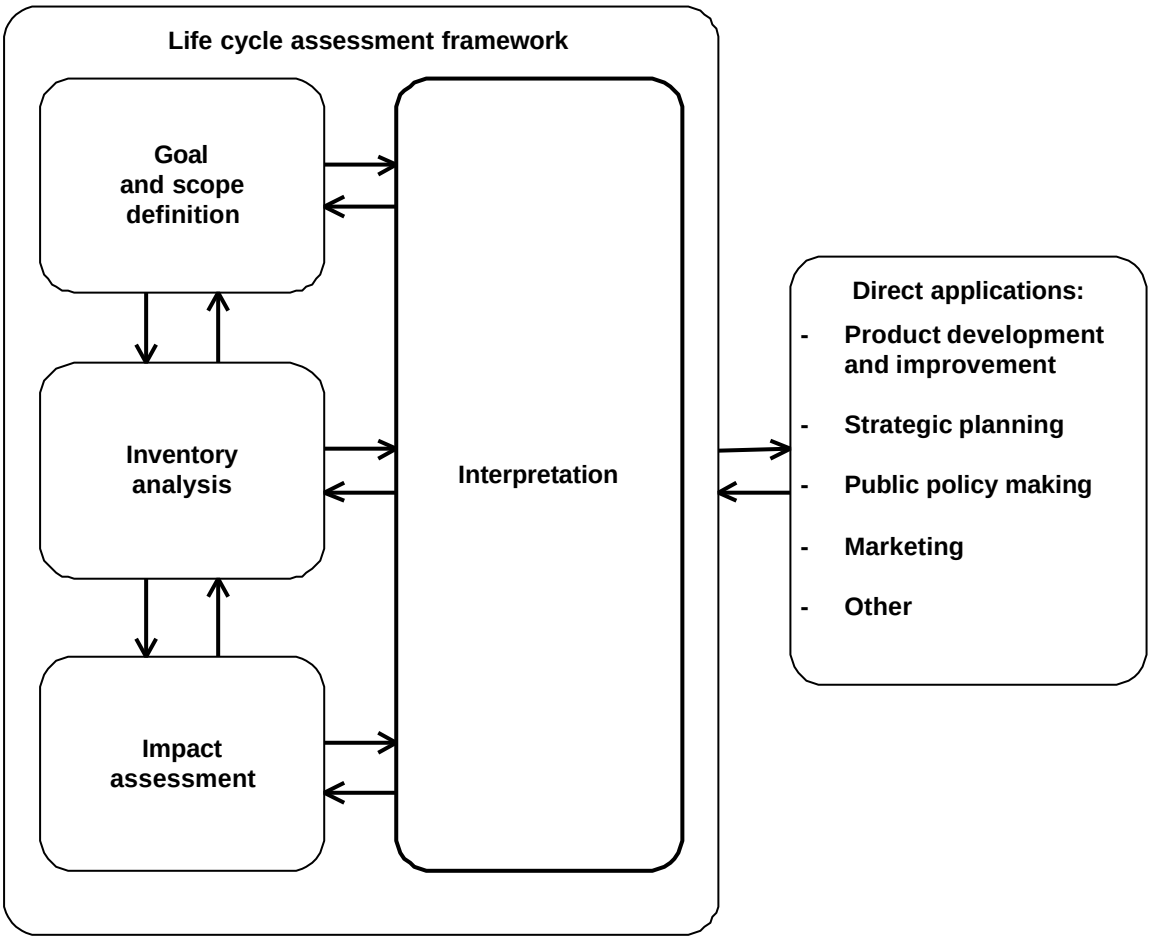


Figure 5.1.1: The Interpretation phase as part of the general methodological framework of LCA (source: ISO 14040, 1997E).

ISO 14043 (2000E) defines Interpretation as “a systematic procedure to identify, qualify, check, and evaluate information from the results of the LCI and/or LCIA of a product system, and present them in order to meet the requirements of the application as described in the goal and scope of the study. Furthermore, Life cycle Interpretation includes communication to give credibility to the results of other LCA phases (namely the LCI and LCIA) in a form that is both comprehensible and useful to the decision maker.” The main aim of Interpretation is to formulate the conclusions that can be drawn from the LCA. In addition, this is the place for reflection on the results of the previous phases of the LCA and on the choices that have been made during the entire process of generating these results. It should be clear, however, that Interpretation, no matter how comprehensive it may be, can never replace an external,

interactive peer review. Such a peer review is strongly recommended unless the study is explicitly for internal use only.

Interpretation as elaborated in this Guide comprises seven steps (see Section 1.4):
- Procedures (no special Section in this volume; see Chapter 1);
- Consistency check (Section 5.2, p. 255);
- Completeness check (Section 5.3, p. 255);
- Contribution analysis (Section 5.4, p. 256);
- Perturbation analysis (Section 5.5, p. 256);
- Sensitivity and uncertainty analysis (Section 5.6, p. 257);
- Conclusions and recommendations (Section 5.7, p. 261).

All these steps should be a regular part of LCA and similar for all levels of sophistication. A further starting point for elaborating these Interpretation steps is ISO 14043 (2000E) with respect to the methodological framework. Here, the ISO recommendations will be further operationalised taking into account the work of SETAC Working Groups and relevant proposals made by other authors. Deviations from ISO will be made only if there are important reasons for doing so.

It is recently that the Interpretation phase was introduced by ISO and it is therefore a topic scarcely referred to in previous LCA literature. The steps of the Interpretation phase will therefore not be discussed according to the previously used format of 'Topic', 'Developments in the last decade', 'Prospects', 'Conclusions' and 'Research recommendations'. As explained in Section 1.5, the step dealing with procedures is not discussed separately in this chapter, but in an integrated fashion, for all phases, in Section 1.3.

One of the main aims of Interpretation is to check the results of the Inventory analysis and of the Impact assessment against the Goal and scope definition of the study. In general terms, it should be asked whether the results actually answer the questions posed in the Goal and whether the answers are within the defined Scope. If, for example, the question relates to a future situation and the data used dates back to the early eighties, the results will not be in line with the Scope of the study. The results of the LCA study are confronted with the Goal and Scope in the following steps of Interpretation:
- Consistency check (Section 5.2, p. 255):
  The assumptions and models used in the LCA should be consistent with the Goal and scope of the study and consistent among the various product systems reviewed.
- Completeness check (Section 5.3, p. 255):
  The parameters describing the system, the data and methodology used in the various phases of the LCA and the results and conclusions of the analysis should all be consistent with the Goal and scope of the study.
- Sensitivity and uncertainty analysis (Section 5.6, p. 257):
  The process data sources (e.g. representativeness with respect to time, space, technology, etc.) and models used, the methodological choices and assumptions made and data reliability should all be consistent with the Goal and the scope of the study.

With respect to reporting on Interpretation, ISO 14043 states: "The report shall give a complete and unbiased account of the study, as detailed in ISO 14040. In reporting the Interpretation phase, full transparency in terms of value-choices, rationales and expert judgements made shall be strictly observed."

This implies that the general requirements of ISO 14040, clause 6 (see Chapter 2) also apply here.

For Interpretation the following reporting issues can be derived from Lindfors *et al.* (1995a):
- The results of the study should be discussed in relation to aspects that may influence the results.
- The results of the study should be discussed in relation to earlier, related studies.
- Conclusions drawn shall be justified by material presented in the report and be based on the whole report.
- When results and conclusions are presented, aspects that may influence the results shall be mentioned.
- Biographies and/or current positions of members of the reference panel or review group shall be reported, as relevant.
- A short resumé of the discussions in the reference panel shall be given, with a focus on conflicting views, OR
- A report from the reviewer(s) on the critical review (see Section 1.3), stakeholder review or validation, OR

– A statement that an external validation or review process has not been carried out, including a justification of that decision (e.g. since stakeholders have been involved in the conduct of the study).

5.2 Consistency check

The aim of the consistency check is to determine whether the assumptions, methods, models and data are consistent with Goal and scope of the LCA study, in terms both of the chain embodied in individual product life cycles and among the products compared. ISO provides the following examples of inconsistencies (ISO 14043, 2000E):

– "differences in *data sources*, e.g. Option A is based on literature, whereas Option B is based on primary data;
– differences in *data accuracy*, e.g. for Option A a very detailed process tree and process description is available, whereas Option B is described as a cumulated black-box system;
– differences in *technology coverage*, e.g. data for Option A is based on experimental process (e.g. new catalyst with higher process efficiency on a pilot plant level), whereas data for Option B are based on existing large-scale technology;
– differences with *time-related coverage*, e.g. data for Option A describe a recently developed technology, whereas Option B is described by a technology mix, including both recently built and old plants;
– differences in *data age*, e.g. data for Option A are 5-year old primary data, whereas data for Option B are recently collected;
– differences in *geographical coverage*, e.g. data for Option A describe a representative European technology mix, whereas Option B describes one European Union member country with a high-level environmental protection policy, or one single plant".

A final example is differences in functions the two products or options perform. These differences should either be justified or corrected. The influence of differences that cannot be corrected or justified on the results and conclusions should be determined in a sensitivity analysis (see Section 5.6).

Involving technological and other experts and comparing the results of the study with those of previous studies on related subjects may be very useful in the consistency check.

5.3 Completeness check

One way to spot incomplete or even erroneous data is to have an expert examine the results of the LCA and the way the results were generated. An LCA expert can examine the methodology used in the various phases of the LCA, and the results and conclusions of the analysis, all in relation to the Goal and scope of the study. Besides the LCA expert, technical experts should also have examine the parameters used to describe product systems and the quantitative data employed. The LCA expert may be able to uncover assumptions or methodological choices that are incompatible with the Goal and scope of the LCA, while technical experts may spot unexpected, missing or erroneous emissions, economical inflows and outflows or product characteristics.

Another way in which incomplete or erroneous data can be uncovered is to compare the study with other, similar studies. Again one should focus on the parameters employed to describe the system, the methodologies applied in the various different phases of the study, the data used and the results and conclusions of the analysis, all in relation to the Goal and scope of the study. When comparing two LCA studies great care should be taken that the respective Goal and scope definitions are truly congruent.

The above recommendations should be implemented in iteration with the steps 'Interventions ' (4.3.17), 'Economic flows not followed to system boundary' (4.3.18) and 'Cut-off and data estimation' (3.8). Particular attention should be paid to comparisons between alternative product systems. If there is significant variation in the completeness of the data sets of the respective alternatives, the potential influence of this difference should be estimated (using contribution analysis, perturbation analysis or sensitivity analysis, for example; see the following sections).

5.4 Contribution analysis

The aim of contribution analysis is to establish the contribution to the overall LCA result of various identifiable elements and parameters. In the production of 1000 litres of milk, for example, the shares might be calculated of:

- individual processes, e.g. pasteurising;
- a group of processes, e.g. refrigerated storage;
- a life-cycle stage, e.g. (dairy) production;
- the packaging, e.g. the bottle;
- an intervention, e.g. SO_2 emissions.

In other cases, contribution analysis may focus on specific product properties, such as the power consumption of a refrigerator.

Contributions to the overall LCA result can be calculated at different levels:

- at the level of the weighting results;
- at the level of the indicator results and/or normalised indicator results;
- at the level of inventory results, e.g. emissions or resource extractions.

Although the questions posed in the contribution analysis may seem quite straightforward, the implications may be more intricate. An example is the question: 'What is the contribution of the power consumption of a refrigerator to the total score on climate change ?' The approach would normally be to calculate this consumption in relation to the functional unit, go back a step in the flow diagram and calculate how much CO_2 is emitted during power generation and, finally, to calculate the contribution of this CO_2 to the total score on climate change. However, this is only part of the story: what about the emissions of the tankers transporting the oil to the power plant, for example? The simple question we began with thus has answers at different levels: the *zero-order* answer would take into account only the emissions occurring during the process 'refrigerator use'. The CO_2 emissions in this process are zero, however. The *first-order* answer to the question would take into account the emissions at the power plant, while the *second-order* would also cover the emissions of tankers and the *third-order* answer would even consider the emissions occurring during the extraction of the fossil fuels, etc. Normally speaking, in a contribution analysis a rather arbitrary number of steps are taken back in the flow diagram. The contribution analysis establishes the extent to which a particular environmental intervention contributes to a certain environmental score. The contributions of all the environmental interventions associated with a particular process can of course also be summed to calculate the zero-order contribution of this process to the overall LCA result.

However, this means ignoring all other linkages to other economic processes.

In most cases there will be a second question behind the one formulated above, of the form: 'Which intervention, economic flow or process can we best change in order to reduce the climate change score of the product system'. This type of question, typical of analysis of improvement options ('improvement analysis'), can only be properly answered if the linkages among the different processes are covered right up to the highest order. Since normal, zero-order contribution analysis does not take such interlinkage into account it cannot be used to answer this type of question. For this purpose perturbation analysis can be used, the topic of the next section.

Contribution analysis can be used to focus the sensitivity analysis on those variables, flows, etc. having greatest influence on the LCA result. Due caution should then be exercised, however. Flows that have been underestimated or disregarded entirely, for example, may initially appear to make little or no contribution to the overall results of the study. Once corrected or included, though, such data may nonetheless affect results significantly. These 'false negatives' will therefore not show up in the contribution analysis as key issues for a sensitivity analysis.

5.5 Perturbation analysis

In a perturbation analysis[1, 1] the effects are studied of small changes in the parameters that describe the system on the overall results of an LCA (Heijungs, 1992; Heijungs, 1994). The effects of these small

[1] This type of analysis is referred to as marginal analysis by Heijungs *et al.* (1992). Using this term here would lead to confusion, however, because of the use of the term 'marginal' in a different context in the Goal and scope definition and Inventory analysis.

changes are calculated simultaneously for all the flows of a system, i.e. economic flows and environmental interventions. The analysis may be performed at different levels of aggregation: inventory table, indicator results, normalised indicator results or weighting results. All the factors used to calculate the aggregated result are included in the perturbation analysis. For example, if the analysis is performed at the level of weighting results, the characterisation factors, the normalisation factors and the weighting factors are all included. The outcome of the perturbation analysis takes the form of a list of multiplication factors or multipliers. The values of these multipliers may range from minus infinity to plus infinity for economic flows and from -1 to 1 for environmental interventions. In fact the results for environmental interventions are identical to the results of the contribution analysis except for their being expressed as multipliers rather than percentages: a multiplier of 0.98 for an environmental intervention in the perturbation analysis would correspond to a contribution of 98% in the contribution analysis. The main difference between the contribution analysis and the perturbation analysis lies in the fact that the latter is concerned not only with environmental interventions but also with economic flows. This is even more important because of the fact that multipliers of economic flows may exceed unity if internal loops are involved. A multiplier of 4.5, for example, would indicate that an 1% increase in this flow would lead to a 4.5% change in the selected inventory or indicator results. It is, of course, clear that this kind of magnification is very important in the context of improvement analysis and in a sensitivity analysis aiming at detecting important issues and refining the LCA study in question.

The result of the perturbation analysis is normally a list of processes or flows with associated multiplication factors in decreasing order of significance for a specific type of result (e.g. CO_2 emission, indicator result for climate change, etc.). Although the mathematics can be quite complicated, the perturbation analysis is relatively easy to implement if a matrix type of calculation method is used. Besides use in improvement analysis, it can help to focus the sensitivity analysis on those variables and (modeling) choices of greatest influence on the results of the study. In this respect, it can significantly reduce the effort required for gathering uncertainty data, because it identifies which data items are crucial for uncertainty analysis. In interpreting the results of the perturbation analysis due caution should again be exercised, however; as with the contribution analysis described above, 'false negative' issues will not be identified (*cf*. Section 5.4).

5.6 Sensitivity and uncertainty analysis

If LCA is to be usefully employed as a decision-making tool, the robustness of the results must be clear. This step of the Interpretation phase assesses the influence on results of variations in process data, (model) choices and other variables. In the sensitivity analysis these changes are deliberately introduced in order to establish the robustness of the results with regard to these variations. In the uncertainty analysis empirical data on the uncertainty ranges of specific data are used to calculate the total error range of the results.

In order to assess the robustness of the results information is required on both their validity and their reliability, distinguished as follows:

– with *validity* the question to be answered is whether the results are based on sound reasoning or, in LCA, whether the appropriate (e.g. correct representativeness in space, time, technology etc.) process data (sources) and models have been used and whether the appropriate methodological choices and assumptions have been made, all in relation to the Goal and scope of the study. If any controversial choices have been made, the influence of these choices on the results of the study should be assessed. Variability, as introduced by others in the context of LCA (e.g. Huijbregts,

[1] It is debatable whether perturbation analysis should constitute a separate step or be included in sensitivity analysis

(5.6). In perturbation analysis, so it can be argued, the data used to describe the system are altered. However, these small changes are used only to assess which processes or flows are most important in the system as modeled. Thus, perturbation analysis can be said to assess intrinsic system sensitivity rather than the effects of estimated uncertainties in variables or modeling choices.

1998a; Huijbregts, 1998b, Hertwich, 1999), is regarded as being subsumed under validity. When discussing validity it should be borne in mind that assessment thereof is closely related to basic choices (see Section 1.2.2.3);
- with *reliability* the question to be answered is whether the parameters and data used are likely to be true or correct, i.e. whether they are to be trusted or believed. The issue of reliability is closely related to that of data quality within the context of LCI (Van den Berg *et al.*, 1999).

As indicated earlier, both data (sources) and (modeling) choices should, in principle, be subjected to an analysis to ascertain their validity and reliability. Some issues (previously referred to as 'issues for Interpretation') to be subjected to sensitivity analyses may have been identified in previous sections under the specific headings of Goal and scope definition, Inventory analysis and Impact assessment. Additional issues can be derived from the results of the contribution analysis and the perturbation analysis. In this step a final selection of issues to be subjected to sensitivity analysis is first made. The number of issues selected may depend on the level of sophistication of the LCA.

In the context of sensitivity and uncertainty analyses, a number of technical terms occur again and again. In this textbox, the most important of these are explained. Note, however, that terminology is not uniform, so that other meanings for the same terms and other terms for the same concept may be encountered in literature.

A basic distinction must be made between accuracy and precision. Data may be precise but inaccurate, for instance when use is made of a high-precision clock which has not been adjusted to local time. Data may also be accurate but imprecise, for instance when a sundial is used.

The result of a calculation is sensitive to several sources of uncertainty. We mention uncertainties in the data, for instance when there are several different measurements or estimates of an emission, and uncertainties in the model, for instance due to essentially arbitrary decisions relating to system boundaries, allocation and so on. Data uncertainty may arise because data are in themselves variable, for instance different in London and in Paris, or on Monday and Sunday, or the data may be the outcome of a stochastic process. The former is referred to as variability, the latter leads to sampling error. In addition, data may be measured incorrectly; we can then distinguish random errors from systematic errors. In general, random errors lead to inaccurate data, and systematic errors to imprecise data.

Model uncertainty leads to doubts regarding the validity of a result, data uncertainty to doubts on its reliability. Even when results are highly imprecise or inaccurate, they may still be robust. For instance, if product X remains preferable to product Y even when the absolute difference varies wildly under sensitivity analyses, the ranking of these two products is said to be robust.

In a simplified LCA the sensitivity and uncertainty analyses are confined to a limited checklist. This checklist includes those (model) choices known to be of major influence on the results of the study:
- product system specification (product composition, trip rates, recycling rates, life span, etc.);
- allocation rules;
- characterisation method;
- weighting method.

The consequences of altering the choices vis-à-vis these issues on the results of the LCA should be calculated.
Results will always be particularly sensitive to changes in economic flows in close proximity to the functional unit. Thus, the product system specification probably constitutes the most important data for the results of the study. This data should therefore be checked carefully and subjected to a sensitivity analysis. With regard to data uncertainties, in a simplified LCA one should focus on those processes and flows already identified as being of key importance in the contribution and/or perturbation analysis. The data on these processes and flows should be checked and a partial sensitivity analysis performed by varying these data.

In a detailed LCA the sensitivity and uncertainty analysis is also confined to a checklist, but a more comprehensive one (based on ISO 14043, 2000E):
- allocation rules;
- cut-off criteria;
- boundary setting and system definition;

– process data;
– characterisation method and data;
– normalisation data;
– weighting method and data.

With regard to data uncertainties, in a detailed LCA one should also focus on those processes and flows identified in the contribution and/or perturbation analysis as being most important. The data on these processes and flows should be checked and a partial sensitivity analysis performed by varying these data.

As an option for extension the consequences of the (modeling) choices and data uncertainties may be quantified in more detail and partial uncertainty analyses performed. First of all the consequences of different (modeling) choices will have to be assessed, as in detailed LCA. Besides the effects of (modeling) choices, however, the effects of uncertainties in the process data should also be determined.

In the most general terms, the question to be answered by an LCA is whether two product systems differ significantly at the level of inventory results, (normalised) indicator results or weighting results. If an LCA is performed without any uncertainty or sensitivity analysis being undertaken, the results take the form of two scores: one for product system A and one for system B. A comparison of the two systems may indicate that the environmental score (e,g, emission of CO_2) of B is greater than that of A, for example, and the conclusion drawn that A is better than B from an environmental angle.

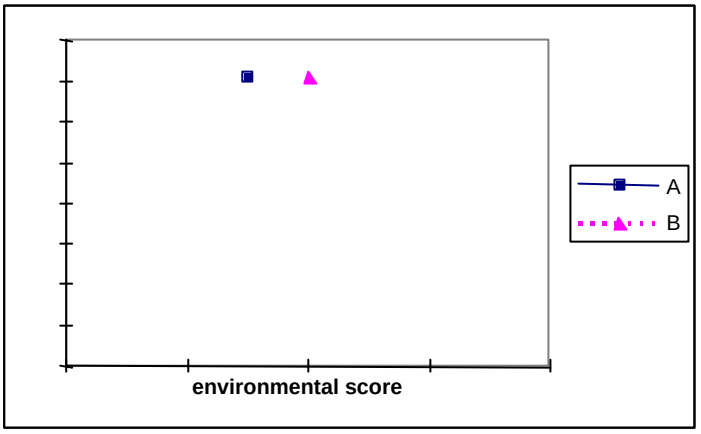


Figure 5.6.1: Environmental scores for two hypothetical systems A and B.

However, the question should be whether there is indeed a significant difference between the CO_2 emissions of the two product systems or whether it is merely an artefact of the uncertainties in the values of the system parameters and in (modeling) choices made. As is evident from the above, the robustness of the end results of an LCA is subject to a large number of validity and reliability issues and the value of the result is not a single point but some kind of probability distribution.

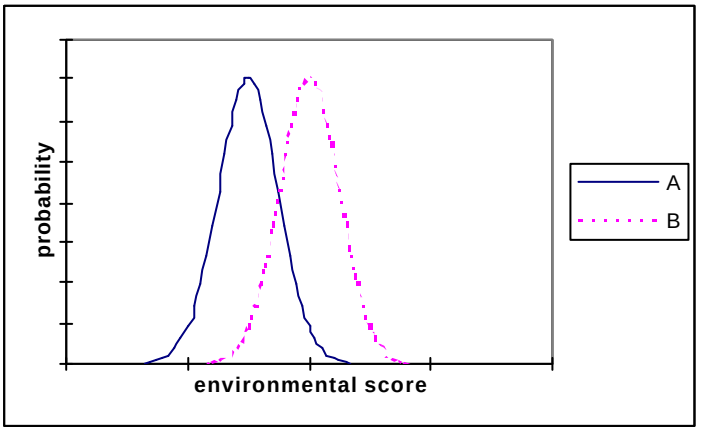


Figure 5.6.2: Probability distribution of environmental scores for two hypothetical systems A and B.

Thus, in order to judge this robustness, the combined influence of all the issues mentioned above must be gauged. Several methods have been proposed for this purpose, of which three are discussed here:

1. calculation of extreme values;
2. formal statistics: uncertainty propagation;
3. empirical statistics: Monte Carlo simulation.

1. Calculation of extreme values

One apparently simple approach is to calculate extreme values. In this calculation the upper and lower values of each parameter are combined to find the upper and lower values of the end result. Heijungs (1996) shows that due to the inherent complexity of an LCA (e.g. the presence of feedback loops) the extremes of the result cannot be predicted by intuition. He also shows that this implies the need to test every combination of upper and lower values and that for an average LCA this task would take a modern PC longer than the current age of the universe. This kind of uncertainty analysis is therefore not of much use in most LCAs.

2. Formal statistics: uncertainty propagation

Heijungs (1996) proposes a formal solution using a standard statistical method: propagation of uncertainties. In this case one starts not by determining the upper and lower values of a given parameter but by assuming a particular distribution of the parameter values. If a normal distribution is used, the mean and standard deviation of the parameters must be calculated. Although the mathematics is complicated, in itself the method is relatively simple to implement in automated calculation procedures, that is, if matrix calculation is used. This formal statistical approach yields such statements as: with a 95% certainty interval, the CO_2 emission of product system A is greater than that of product system B.

3. Empirical statistics: Monte Carlo simulation

Another technique that can be used to avoid the problems associated with calculation of extreme values is stochastic modeling. This technique can be performed with the aid of a Monte Carlo or Latin Hypercube simulation (Huijbregts, 1998a and 1998b). In both types of simulation a predefined, limited number of combinations (typically 10,000) of random parameters, restricted by their uncertainty distribution, is used to calculate the results. The only difference between a Monte Carlo and a Latin Hypercube simulation is that in the former the uncertainty distribution of each parameter must be specified while in the latter the uncertainty distribution is segmented into a series of non-overlapping intervals, each having equal probability. One advantage of stochastic modeling is that, in contrast to formal statistic methods, it is relatively easy to employ a variety of parameter distributions, such as uniform, triangular, normal and log normal. The result of this type of analysis is a frequency chart of possible outcomes.

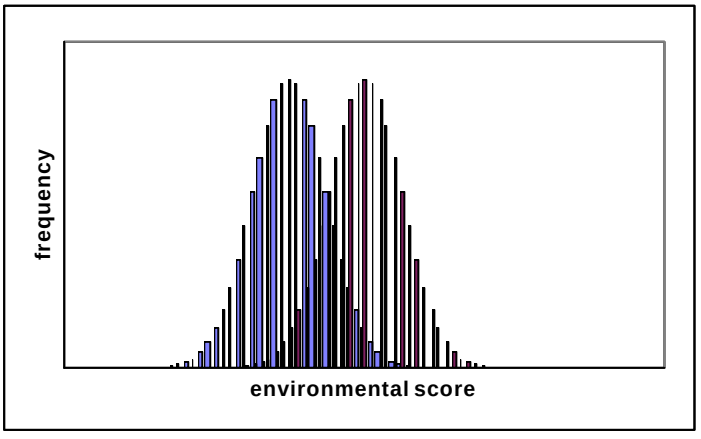


Figure 5.6.3: Frequency distribution of environmental scores for two hypothetical systems.

Once a frequency chart has been generated, the same statistical methods used in the aforementioned formal statistics approach can be used to assess whether or not two product systems differ significantly.

In all three methods described, information is required on uncertainties in parameter values. This requires a major additional effort, coming on top of actual data collection, in itself already one of the most time-consuming tasks of LCA. There are two ways to tackle this problem of availability of data on ranges of uncertainty:

1. use rough estimates of standard deviations or upper and lower bounds instead of real values to obtain some kind of subjective probability distribution;
2. focus data collection efforts on those flows of greatest importance for the study results (as determined by contribution and perturbation analysis): a partial uncertainty analysis.

As already proposed by Heijungs (1996) and Huijbregts (1998a, 1998b), focusing on key parameters would greatly simplify matters. One way to rank parameters in order of importance for the study results is to use perturbation analysis (see Section 5.5). This could greatly reduce the amount of information required for a sensitivity analysis. However, the results of both options (i.e. 1 and 2) should be treated with great caution: by introducing subjective and/or partial probabilities one may also introduce an erroneous notion of the probability of the results, while only partial or estimated uncertainty data have been used.

Another important issue is that both in the formal statistical approach and in the standard Monte Carlo simulation the variables are assumed to be independent of one another. In practice this is often not the case; fossil fuel inputs are closely related to CO_2 emissions, for example. Although relationships among variables can, in principle, be taken into account using co-variances, in practice this will be difficult to implement. If such dependencies are ignored, however, uncertainties will be overestimated.

The approach outlined above can also be used to assess the influence of (modeling) choices. However, current LCA software will not always permit implementation of the required procedures. Such is the case with choices concerning allocation, for instance. In order to assess the influence of adopting a certain allocation procedure it should be possible to introduce this change without having to reorganise the whole data set. Use of aggregated process data can therefore lead to problems during Interpretation.

5.7 Conclusions and recommendations[1]

In this step of the Interpretation conclusions are drawn and recommendations made on the basis of the information gathered in the previous phases of the LCA combined with the results of the previous steps of the Interpretation. In ISO 14043 (2000E) the objective of this step is defined as “to draw conclusions and make recommendations for the intended audience of the LCA or LCI study”.

In general the conclusions of any study should comprise the main results of the study and a discussion of the validity and reliability of those results. In the case of LCA the following items should be included (slightly adapted from ISO 14043).

- a summary of significant issues;
- an evaluation of the methodology and results on the basis of the consistency check, completeness check and the sensitivity and uncertainty analysis;
- the main conclusions as they relate to the Goal and scope of the study, including data quality, predefined assumptions and values, and application-oriented requirements.

Firstly, conclusions should be consistent with the results found and with the original Goal and scope of the study[2], viz. in line with the limitations of the scope, main data and (modeling) choices, and in line with the limitations of the instrument of LCA itself. This implies that the conclusions are only valid for the systems analysed and thus not, automatically, for other similar systems that have not been analysed. For example, the results for 1-litre packaging alternatives for a certain liquid are not valid for similar 1.5-litre packaging alternatives. Separate justification is needed if the conclusions are expanded to other similar systems.

[1] Note that this section does not constitute the overall 'Conclusions and recommendations' for the present Guide. Material relating to this aspect is included under 'Research recommendations' in most of the individual sections of this Guide.

[2] If this is not the case, an explicit statement should be made of the problems encountered and the conclusions formulated in such a way as to reflect these problems in a clear and proper manner.

It also implies that the conclusions are valid for the main data and (modeling) choices made, and not for an altered system with different data and/or (modeling) choices. If, for example, the system boundary is expanded to include more processes earlier in the chain, the conclusions may no longer retain their validity. Finally, formulating conclusions in compliance with the limitations of the LCA instrument implies that conclusions as to the preferred location of a certain industrial facility cannot be drawn on the sole basis of an LCA.

Secondly, the inclusion of the results of the earlier steps of the Interpretation in the formulation of the conclusions is crucial. Performing consistency and completeness checks and sensitivity and uncertainty analyses on data and models is one thing; processing the results of these checks and analyses in the conclusions of a study and formulating recommendations is quite another. There are examples of studies in which sensitivity analyses have been performed, but with absolutely no further processing of the analysis outcomes in the final results.

How, then, should these results be properly taken into account? Inconsistencies, incompleteness and errors should either be corrected or incorporated in the conclusions of the study. The data or parameters to which the conclusions of the study are most sensitive should also be reported. Uncertain data and parameters should be checked and if uncertainties remain, this should be incorporated in the conclusions. One simple but not particularly elegant way of doing so would be to determine for the most dominant data the uncertainty (partial uncertainty analysis) and to add the results of these analyses (see Section 5.6) to a maximum uncertainty range for data. In a similar way the uncertainty could be determined of the most important modeling choices (e.g. allocation models and characterisation models for some impact categories) and the results of these analyses could be added again to a maximum uncertainty range for models. Subsequently, further Interpretation is up to the practitioner and/or decision-maker.

A more elegant solution, which is not yet practically available, is to formulate both data and model uncertainties as input uncertainties (e.g. data: 5 ± 0.5 and models as the probability, say 0.333 on model A, B or C) for a Monte Carlo analysis. In this way all model and data uncertainties are aggregated into a total frequency distribution of the end results of a study. This is, however, not yet feasible and may never be.

Drawing appropriate conclusions is of even greater importance for comparative assertions, in order to minimise opportunistic use of results (‘hired gun’ effect). In this case it will also have to be determined which differences in results are significant in order to be able to conclude that one product alternative is environmentally sounder than another.

It is our conviction that even waterproof Interpretation cannot prevent misuse of results of LCA studies, it can only minimise it. It will always be possible to manipulate results or use results to answer the wrong questions. By providing guidelines or checklists, possible misuse can be minimised but not precluded. Therefore, a peer review process and appropriate procedures are in most cases of crucial importance (see Section 1.3).

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Annex A: Contributors

Besides the authors mentioned on the headpage of this report, a large number of people have contributed to this study and its coordination. It would be impossible to mention all those who have contributed information, at whatever level of detail. We hence limit ourselves to listing and thus acknowledging the members of the steering committee, the think tank, the supervisory committee and the international observers group. Note that referencing of names below does not constitute endorsement or recommendation for use on behalf of the individuals involved.

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Appendix B: Areas of application of LCA[1]

Aksel Roorda (TNO) [2]

Bernhard van der Ven (TNO)

[1] This appendix is a translated and somewhat abbreviated version of a document written in 1997, prior to the start of the present project. It has been included as an appendix to this volume of the Guide because of its more detailed coverage of the actual and potential areas of application of LCA. As far as possible, the terminology of this appendix has been brought in line with that employed in the main text of the Guide.

[2] Currently employed at IWACO, 's Hertogenbosch.

Summary

There are a variety of tools available for assessing the influence of a given 'human activity' on the natural environment. Which is the most appropriate depends on the particular issue being addressed, and the tools are indeed generally classified in terms of their fields of application.

The focus of Life Cycle Assessment is on assessment of a specific, selected system: either a product or a set of processes (a product being conceived of as a particular embodiment of a set of processes). The defining characteristic of such a system is that it addresses a particular demand in society; it fulfils a certain function. The system gains its coherence from the economic supply chain and an LCA accounts for every stage in the life cycle of the system. Actual assessment focuses on the system's impacts on the environment. The chain of cause and effect underlying these environmental impacts is complex, and the assessment parameters employed in LCA are therefore merely *indicators* that stand for *potential* impacts. Many of them are so-called 'stressors', i.e. indicators referring to the start of the causal chain.

Consequently, LCA is eminently suitable for addressing chain-oriented issues. The LCA methodology allows the impacts occurring at each link in a given supply chain to be aggregated, down to the level of relevant 'sub-activities'. The outcome is, of necessity, an *indication* of the potential environmental impact of the product in question.

LCA is less appropriate as a tool for assessing the environmental impacts associated with a specific locality (a specific ecological system). For addressing this kind of issue, tools like risk analysis and environmental impact assessment are far better. Neither is LCA the preferred instrument for evaluating systems in which the temporal aspect of the intervention-impact chain cannot be ignored. A case in point is the landfilling of waste. Not only do activities extend over a certain period of time; the causal chain also varies with time. A third task for which LCA is inappropriate is assessment of systems in which the 'function' comprises a blend of physical and 'affective' aspects.

New applications of LCA are to be sought in the category of complex functions, such as:
- multifunctional systems (including cascades);
- collective functions (including infrastructure);
- clusters of functions (including lifestyles).

In existing fields of application, LCA can be used across a broader range if methodological improvements are made with regard to site-specific aspects and the duration of exposure to emitted substances.

1. Introduction

This appendix outlines the limitations and potential of LCA as a tool for environmental analysis, its relation with other environmental tools and possible new fields of application. The text is a translated and somewhat abbreviated version of a document written in 1997, prior to the start of the present project. It has been included as an appendix to this Part of the Guide because of its more detailed coverage of the actual and potential areas of application of LCA. As far as possible, the terminology of this appendix has been brought in line with that employed in the main text of the Guide.

2. The environmental toolbox

2.1 Introduction

There are a wide range of analytical tools available for inventorying and analysing the environmental aspects of a given object, the most common of which are:

- Risk Assessment (RA);
- Environmental Impact Assessment (EIA);
- Technology Assessment (TA);
- Life Cycle Assessment (LCA);
- Substance Flow Analysis (SFA);
- Environmental Management Systems (EMS).

Which tool or combination of tools is to be used in a particular situation depends on the aim and scope of the study. This section provides a brief review of the individual tools, with the aim of highlighting their similarities and differences. Against this background the potential applicability of LCA is more precisely delineated.

The list of tools distinguished here deviates in several respects from lists in other literature sources, which are frequently longer; see for example (Beck & Bosshart, 1995; Cowell *et al.*, 1997; UNEP, 1996; Udo de Haes & Huppes, 1994; SETAC, 1998). Most of these cases involve downstream links in the chain between 'means' and 'ends'. A case in point is the *environmental audit*. Rather than being a primary *tool*, it is better to consider the environmental audit as just one element of an environmental management system employing a (varying) range of different instruments.

2.2 Review of tools

RISK ASSESSMENT (RA)

The term risk assessment is used for a wide variety of methods concerned with assessing the adverse effects of an activity on human safety and ecosystems. Two main types of risk assessment can be distinguished:

1. accident risk analysis: analysis of the risk of calamity and the attendant human environmental impact;
2. toxicological risk assessment: analysis of the toxicological impacts of 'routine' industrial operations.

1. Accident risk analysis

RA is used mainly in situations where events with a low probability of occurrence are associated with major consequences, such as calamities at chemical or nuclear facilities. The analysis always focuses on the risks that a particular human activity poses to the surrounding area and is therefore always site-

specific. Traditionally, risk assessment has been concerned primarily with assessing risks to personnel and local residents, but today it is usually extended to cover ecological risks to water and land. RA is often undertaken by government agencies, prompted by regulations like the European 'post-Seveso' directive, which obliges firms to demonstrate that the risks posed by a given activity do not exceed certain statutory limits.

2. Toxicological risk assessment

This type of risk assessment is concerned with the effects of 'routine' industrial operations. Here, the probability element is therefore lacking in the analysis, which now focuses on the human toxicological and ecotoxicological impacts of normal, non-calamitous plant operation. This form of risk assessment can be used in site-specific as well as non-specific studies.

As its name indicates, site-specific risk analysis (RA_{ss}) charts the local human and ecological risks of routine industrial operations at a particular facility. RA_{ss} is therefore often used to assess whether risks (concentrations) are below statutory limits. In addition, though, RA can also be used to delineate the causal chain and assess ultimate ecosystem impacts. This more scientifically oriented form of RA focuses more on (local) species extinction and population effects. Although in practice most RAs today cover only toxic aspects, it is becoming increasingly common for other aspects such as noise and desiccation to be included too.

Site-specific risk assessment may be used to assess both existing and projected situations. In the latter case the RA is often prompted by statutory obligations, with the firm having to demonstrate that risks remain within set bounds. In the case of current operations, risk assessment may be performed either by the firm or by other stakeholders with a view to assessing possible harmful impacts on human or environmental health.

Non-site-specific risk assessment (RA_{nss}) is generally concerned with the risks posed by a particular substance or group of substances. It is not then the risks associated with a particular location that are of interest, but the consequences in a given geographical region or country of use of the substance, with all its attendant emissions. This form of RA is often used by national governments to underpin decisions on approving or rejecting new (groups of) chemicals or reducing use of substances in current circulation. SFA data are often employed in these studies.

ENVIRONMENTAL IMPACT ASSESSMENT (EIA)

EIA is used to analyse the environmental consequences of a specific, projected economic activity at a specific geographical location and is designed to assist the decision-making authority in approving major (public or private) projects. Based on the environmental impacts identified in the EIA, the authority can then decide whether or not to approve the project. Consequently, EIAs have both a procedural and an analytical side. The procedural aspects aim to ensure that the EIA dovetails with other relevant legislation, and public participation and expert counsel are therefore also important elements of any EIA.

In terms of analytical substance, an EIA is concerned not only with assessing the risks posed by a given activity, for which purpose a site-specific RA is just as suitable. Many EIAs also examine such issues as land use, waste production and raw materials and energy consumption. In addition, elements of the LCA methodology can be used to run scenario calculations in order to pronounce more reliably on the 'environmental compatibility' of given alternatives. Because an EIA procedure is mandatory for certain categories of scheduled projects, EIAs are initiated by the decision-making authority for a specific site or project.

LIFE CYCLE ASSESSMENT (LCA)

LCA aims to inventory the environmental impacts of a given system or systems fulfilling a particular function in demand in society, which may be delivered by one or more products and/or services. The system is generally made up of a chain of interlinked sub-processes, which may be implemented in different localities at different moments in time. The connecting link between these sub-processes is the economic supply chain.

In assessing the environmental consequences of the system, LCA charts the potential rather than actual impacts, yielding a set of indicators representing the potential environmental impacts at the local, regional or global level. LCA thus allows whole systems to be studied and the associated impact routes to be assessed in an integrated manner. LCA can be used to address a wide range of issues of societal concern, not only by industry, but also by governments and consumer organisations (*cf.* Section 3).

TECHNOLOGY ASSESSMENT (TA)

TA is usually described as a tool for assessing the consequences of introducing a new technology. It is concerned not only with environmental consequences, but also with economic, legislative, employment and other effects (UNEP, 1996). In some cases, it may also extend to ethical and juridical issues. Here, though, we restrict ourselves to environmental aspects.

TA has no formal procedure for examining the environmental performance of a technology. The goal of the particular study will determine whether or not it is site-specific, for example. Depending on the goal, a suitable analysis method is adopted. This may be a form of LCA or a type of analysis geared more to specific activities, such as RA. In short, how TA is elaborated depends very much on the question being addressed, which in turn depends on who is initiating the TA: industry, government or academia.

SUBSTANCE FLOW ANALYSIS (SFA)

SFA can be likened to an accounting system for a particular substance, group of substances or materials within a defined area. That area may be the entire globe, or a restricted geographical area, such as an individual country. The aim of the analysis is to gain insight into the flows, sources and sinks of particular substances within the selected area (UNEP, 1996). SFA is concerned not only with economic flows, but also with flows within the natural environment and with relationships between the two.

This instrument is not aimed primarily at establishing downstream environmental impacts, for which purpose other tools like LCA and RA can be used. Which method is ultimately employed to chart the environmental consequences of a given substance flow depends on the objective of the SFA, the defined geographical area and the object of study (substance, group of substances or material). SFA studies are initiated mainly by government agencies.

ENVIRONMENT MANAGEMENT SYSTEMS (EMS)

EMS are used mainly to inventory and improve the environmental performance of a particular economic activity or system. Environment Management Systems are now standardised in the ISO 14000 series of standards (ISO 14040, 1997E; ISO 14041, 1998E; ISO 14042, 2000E; ISO 14043, 2000E) under which firms satisfying certain criteria can become eligible for certification. Because there are prescribed rules and procedures for EMS activities, there is greater transparency and consistency. This is indeed one of the set requirements for improving environmental performance. The standards apply not only to firms; products and services can also be ISO-certified.

Firms may initiate and implement EMS for a variety of motives:

- to improve their environmental 'image'
- to reduce costs by improved control of the production process
- to improve communications with enforcement agencies.

EMS is not a one-off analysis or project but an activity that forms an integral and continual part of the firm's operations. Its similarity with EIA consists in it having both a procedural and an analytical component. One aspect of the procedure relates to establishing in-house rules and procedures for activities impinging on the environment. The resultant environmental performance is to be monitored by analytical methods. Both internal and external audits are carried out to oversee firms' compliance with their own rules and procedures. The external audits also confirm that the rules and procedures are in conformity with relevant ISO standards. These audit procedures also aim to ensure that there is continual improvement of the EMS, including the firm's actual environmental performance. Audits are clearly one element of an EMS that can help boost environmental performance. EMS also has (procedural) steps with a warning and improvement function. In itself, EMS comprises no specific analytical tools for measuring the environmental performance of a firm or product, but makes use of the available toolbox as appropriate.

2.3 Comparison of tools

While the tools differ in many respects there are many overlaps, some quite sizeable. How they relate and compare can be examined by categorising them according to relevant aspects. The SETAC Working Group on Conceptually Related Programmes has drawn up a framework for this purpose that

distinguishes between concepts (such as clean technology, design for environment and industrial ecology) and tools. Tools, for their part, rely on data (SETAC, 1998).
The main features of the various analytical tools are summarised in Table B1, in a less elaborate version of the SETAC scheme, highlighting their differences and similarities.

Table B1: Environmental tools: principal features

Tool	primary economic object	orientation to chain	result	spatial differentiation	temporal aspects	used in
LCA	function	entire life cycle	analysis	global	no	EIA, TA, EMS
RA	project, firm	none	analysis	site-specific	yes	EIA, TA, EMS
RA_{ss}	project, firm	restricted to site	analysis	site-specific	yes	EIA, TA, EMS
RA_{nss}	substance	restricted to defined area	analysis	regional	yes	TA
MER	project	restricted to site, but possible	procedure / analysis	site	yes	(EMS)
SFA	substance	restricted to defined area	analysis	defined area	possible	
TA	technology	possible	analysis		possible	EMS
MMS	firm/function	possible	procedure / analysis	site/function	possible	

3. LCA fields of application: a closer look

3.1 Introduction

This section looks more closely at the (potential) range of application of LCA. The topic can be approached from a variety of angles. In the literature the most common approach is to compare methods, and the table in the previous section is a similar attempt. A perspective less frequently adopted is to describe applicability with reference to the kinds of *issues* actually arising in society. In tandem with an analysis of the strengths and weaknesses of LCA, this kind of review may shed greater light on the contours of (potential) applicability. This is a different perspective from that adopted by (Beck & Bosshart, 1995), who distinguish the following 'activities': services, firm, project, technology and economy. Although these categories are termed 'possible areas of application', they provide little depth of analytical resolution.

3.2 Features of LCA

As a concept and an instrument, Life Cycle Assessment has been described extensively in numerous publications (*cf.* this Guide). On the assumption that the reader is more or less conversant with the substance thereof, this section outlines the principal features of LCA with the aim of examining the *limits* of this particular environmental tool. This is the question that must be answered to establish when LCA can be used and when it cannot. Our point of departure is that every tool has two characteristic aspects[1]: in what situations can the tool be used, and what does it measure?

[1] Beck & Bosshart (1995) provide a more exhaustive list.

1. OBJECT OF STUDY: A SYSTEM CONSISTING OF A DISCRETE SET OF COHERENT ACTIVITIES[1]

The object of study of LCA is a selected system, either a product or a set of processes (a product being conceived of as a particular embodiment of a set of processes). The defining characteristic of such a system is that it addresses a particular demand in society; it fulfils a certain function. The system gains its coherence from the economic supply chain and an LCA accounts for every stage in the life cycle of the system.

The LCA assessment procedure may focus on either a single system or several system alternatives. In the first case, the main problems associated with the selected system are identified. In the latter type of 'comparative' LCA the system of interest is ranked against the selected alternatives. The selected system or systems are modeled as a set of unit processes, described in terms of both economic and environmental inputs and outputs. The object of study is thus a *model* of reality rather than reality itself. The essential issue is whether the model possesses sufficient discriminating power to analyse the consequences of intentional changes in inputs. For example, if a comparison is being made between two paint systems, differing in their performance with respect to emissions and lifetime, the model must (at least) be able to sensitive to variations in these two parameters. A distinction should be made here between system variables and stochastic variables. In the given example, system variables include paint composition, method of application and maintenance requirements, while stochastic variables might include the energy model and transport model employed in the particular LCA' In comparative LCAs it is sufficient that the system variables be adequately discriminated in the model, assuming that the stochastic variables have the same effect on all systems (or are not amenable to control, and thus irrelevant in a decision-support context).
The systems being compared must deliver the same function or functions. If this is not explicitly the case one or more systems may have to be adjusted accordingly, in the process defining the 'functional unit' to be used in the LCA. A basic issue, however, is whether it is indeed always feasible to redefine systems such that they provide an 'identical' function or functions. In LCAs this process of adjustment is generally based on technical functions, with any differences in 'affective' functions being ignored[2].

2. MEASURED VARIABLES: POTENTIAL ENVIRONMENTAL IMPACTS

LCA assesses the environmental consequences of the system(s) under study. The chain of cause and effect underlying these environmental impacts is complex, and the assessment parameters employed in LCA are therefore merely *indicators* that provide an indication of *potential* impacts. Many of them are so-called 'stressors', i.e. indicators referring to the start of the causal chain.

The notion of potential impacts has both advantages and drawbacks. The principal advantage is that LCA provides a quantitative relationship between the 'normative values' arising in society and the overall impact of the environmental interventions attributable to the system(s) in question and does so in a manner that is transparent. The drawback is that this relationship is established at an extremely high level of aggregation and that consequently the *actual* environmental impacts cannot be assessed.

Table B2, below, elaborates these two characteristics in the form of a summary analysis of the strengths and weaknesses of LCA, providing an initial indication of preferred and non-preferred areas of possible application.

[1] Our perspective is 'object-oriented' as opposed to 'structurally oriented', as described by (Heijungs,1997a). In the latter approach the tool is broken down into its constituent parts, from which the range of application then follows.
[2] Cf. Cowell *et al*. (1997) who distinguish a scientific, business and social 'lobe' in the decision-making process; 'affective value' is then part of the social lobe.

Table B2: Strengths and weaknesses of LCA

Feature		Strength	Weakness
1.	Function-oriented	Matches marketplace reality.	Leads to allocation problems with multifunctional activities. Also doubtful whether compared functions are truly identical; see 5.
2.	Potential environmental effect	See below	Relationship with actual impacts is unclear.
3.	Burdens rather than concentrations	In combination with 2 and 3, permits activities at different sites to be related.	Relationship with actual impacts is unclear. Interpretation problems arise with emissions causing near-background concentrations and having a non-linear relationship between concentration and toxicity (especially relevant for nutrients like Zn).
4.	Non-site-specific (in context of characterisation)	See 3	See 2.
5.	Substance-oriented	Properties are measurable and quantifiable.	Social, economic, aesthetic and ethical aspects of the function are ignored.

Features 2, 3 and 4 are interconnected and form the heart of LCA. In interpreting the table, the following should be borne in mind:

- The table is not exhaustive, but restricted to principal issues.
- A number of weaknesses are related to *current* methodology. In some cases methodological refinements and improvements are certainly conceivable. One example would be introduction of site-specific aspects in the context of characterisation.
- The term 'site' in the expression 'non-site-specific' should be understood as referring to Impact assessment rather than to the process description(s) of the system(s) under study. Depending on the goal of the assessment, activities may certainly be very 'site-specific'.
- The table makes no explicit mention of the 'time problem', which can be regarded as resulting from the fundamental uncertainty of the future rather than constituting an inherent inadequacy of the LCA method itself; *cf.* (Udo de Haes & Huppes, 1994). This holds for description of the supply chain (technology) as a function of time as well as for Impact assessment. With respect to the latter a distinction must be made between the time-dependence of normative values, on which Impact assessment is based, the occurrence of environmental impacts as a function of time, and the emissions (etc.) themselves as they vary in time. The last of these is covered by feature 1 and is described in Table 2, above.
- The issue of how well the model must match reality to answer the questions of interest has likewise been ignored above, although this is obviously of immediate relevance for the possible range of application of LCA. Nonetheless, modeling should be regarded in the present context as a separate issue rather than as part and parcel of LCA as a tool, for the use of models is common to many environmental instruments and is therefore an issue to be addressed prior to evaluation of LCA as one of the possible options in the toolbox.

3.3 Issues of concern

It will be apparent that we do not intend here to provide a comprehensive review of all the possible issues of concern in a given society. Where do we draw the line then? In Section 3.1 it was argued that LCA's potential range of application could be staked out by considering how well its key features are suited to tackling the kinds of issues actually arising in society. In this section we attempt to characterise these to the extent that they relate to the object domain examined in the previous section.
As we have seen, LCA is concerned with the economic supply chain. The domain of study is defined by the set of economic activities directed towards the transformation and transfer of products and the

transfer of financial values; in short, by the economic network. Figure B1 provides a schematic illustration.
The sum total of economic supply chains can be considered as a network of economic activities, comparable to the input-output tables of the national accounts, for example. It is generally flows of substances or money that are made visible in this kind of network. At the same time, though, such models can provide a basis for localising stakeholders and the issues of concern in their respective spheres. There are three main categories of stakeholders: producers, consumers and government.

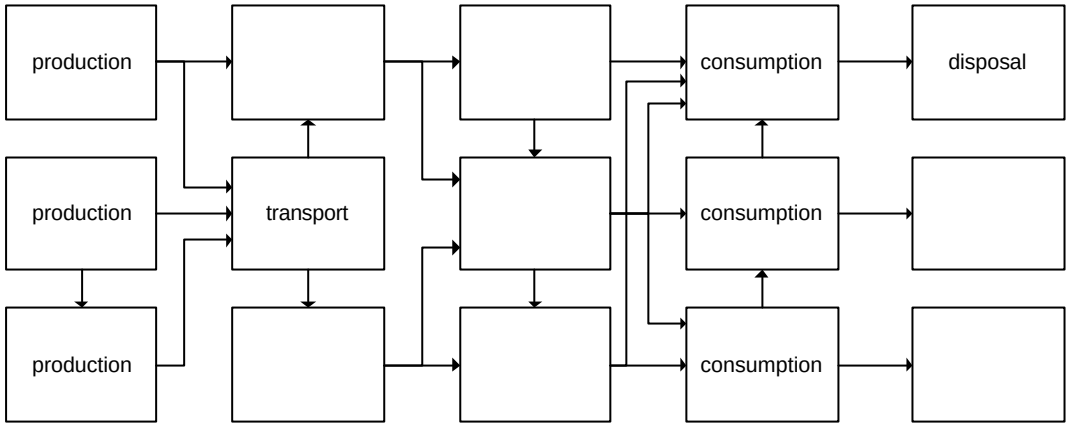


Figure B1: Economic network

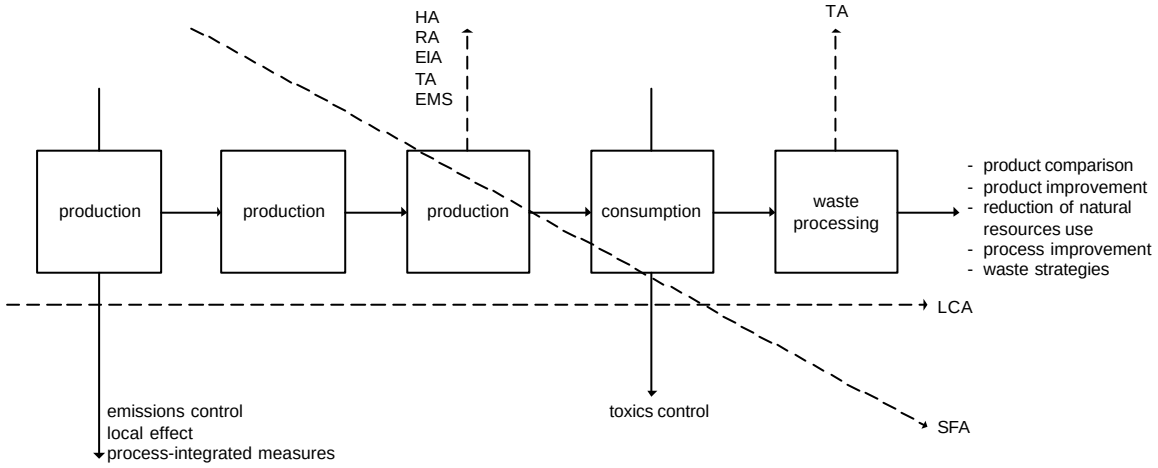


Figure B2: Issues and tools in relation to the production chain (for explanation, see text)

From the same perspective of the supply chain, Figure B2 highlights the issues of concern, and thus areas of potential application. These are indicated by a solid line and fall into two categories: the classic issues, tied to a single step in the chain (in the lower part of figure) and those relating to the chain as a whole (on the right). The various analytical tools are indicated by a dashed line, which in the case of SFA is diagonal to indicate its intermediate status, for it is concerned with more than a single production chain.
The ‘transverse’ issues perpendicular to the chain are familiar. Is the facility in question operating in compliance with discharge permits and standards, and are operations compatible with the local environment? Another focal area is local optimisation (process-integrated measures at a single firm or within a local cluster of firms, e.g. an industrial estate). Depending on the time horizon, these issues may be approached descriptively (permits, annual environmental reports) or strategically (policy decisions, planning). With these ‘transverse’ issues, the questions addressed by the firm and by government agencies are essentially the same, but the interests of the two parties differ.

Corresponding to the producers' issue of 'emission control', consumers are concerned with emissions during the use phase and the associated health risks. These latter issues are translated by government into product standards and regulations for use.

Besides these 'transverse' issues there are also 'longitudinal' issues, relating in principle to the supply chain as a whole. Although this second type of issue may be relevant for all stakeholder categories, here it is always the government that makes the first move. The issues of potential concern to producers and/or consumers are now pro-actively addressed by the decision-making authority.

An important characteristic of the 'longitudinal' issues is their warning function. Given a product and its particular supply chain, the question now essentially addressed is where the main problems are located. These are the precise areas where (environmental) improvements are feasible and the next step, then, is to initiate specific programmes to implement such changes. In physical terms these changes will be identical to those arising directly from the 'transverse' frame of issues.

This brief description does not break down these issues according to stakeholder interests; on this point the reader is referred to (Cowell *et al*., 1997). Here we assume that all the issues (stakeholder-dependent) can be assigned to one of the indicated categories. Table B3 provides further examples (without being exhaustive).

Table B3: Examples of 'longitudinal' and 'transverse' issues

	type of issue	examples
longitudinal issues	product comparison	- which product alternative is preferable? (e.g. packaging) - underpinning of product policy - eco-labeling
	product improvement	- new product design - optimisation of environmental performance along the chain
	reduced raw materials use	- design of new product, process, function
	process improvement	- technological optimisation (e.g. which energy sources?)
	waste strategies	- what combination of processing technology and recycling is best?
transverse issues	emissions control	- how can a firm satisfy new operational standards? (e.g. stricter permit criteria)
	local impacts	- choice of new site, or extension of current operations (e.g. Schiphol Airport)
	process-integrated measures	- how can a firm satisfy criteria to limit on-site environmental impacts? (e.g. environmental care programmes)
	toxics control	- how can products meet government and consumer health criteria? (e.g. Cd in toys)

The situation can be summarised as follows:

> Society as a whole can be conceived as consisting of a network of activities. For each activity and each chain of activities environmental criteria can be established. The first case involves specifically located objects, and the environmental issues arising relate to the activity and location in question. The level of aggregation at which the environmental issues can be addressed is the substance level. From the chain perspective, in contrast, it is the function of the chain that is the object of study, and the overall environmental burden accruing to that function is determined by the sum total of underlying activities. In this case the analysis aggregates at the 'indicator' level, highlighting the activities causing the greatest environmental burden. Actual developments to reduce the environmental burden take place at the level of individual activities.

3.4 Demarcating the scope of LCA

In the previous section we described the main features of LCA, leading to a preferential area of application. A characteristic 'buzzword' for that area is 'longitudinal': whenever it is the supply chain that is the domain of study, LCA is an appropriate tool. Table B4 elaborates, with reference to keywords.

Table B4: Applicability of LCA (keywords)

stakeholder	topic
government	product policy
	prioritisation of 'target-group policy'
	general technological reconnaissance
	prioritisation of environmental technologies
industry	product comparison
	benchmarking (multi-criteria)
	product development
	process development
consumers	product comparison / information

In themselves, the keywords in the table are generic terms referring to entire categories. The idea is that they cover the whole field of 'longitudinal' issues, as expressed by the three stakeholders[1]. Some sources mention additional applications, for example OECD (1995), UNEP (1996) and Beck & Bosshart (1995):
- for eco-labeling (in this study regarded as a type of product comparison);
- for environmental audits (subsumed under 'Environmental Management Systems');
- for negotiations (not a primary field of application, but a downstream link in the policy chain);
- for product information (ditto).

4. New fields of LCA application

LCA applications will always remain restricted to the domain referred to in Section 3 as that of the 'longitudinal' issues. This is indeed the strength of LCA in comparison with other tools. The analytical focus of many of the familiar applications of LCA has been on relatively simple and physically identifiable functions. The 'longitudinal' playing field is larger, though. New[2] areas of application are formed by more complex functions, such as:
- multifunctional systems;
- collective functions (defence, infrastructure, etc.);
- coherent clusters of functions (lifestyles).

We now consider a number of examples from these new areas, providing suggestions for possible implementation.

4.1 Cascade systems

A cascade is a good example of a multifunctional system. In a cascade system there is a substance flow that fulfils multiple functions consecutively. The principle is illustrated in Figure B3.

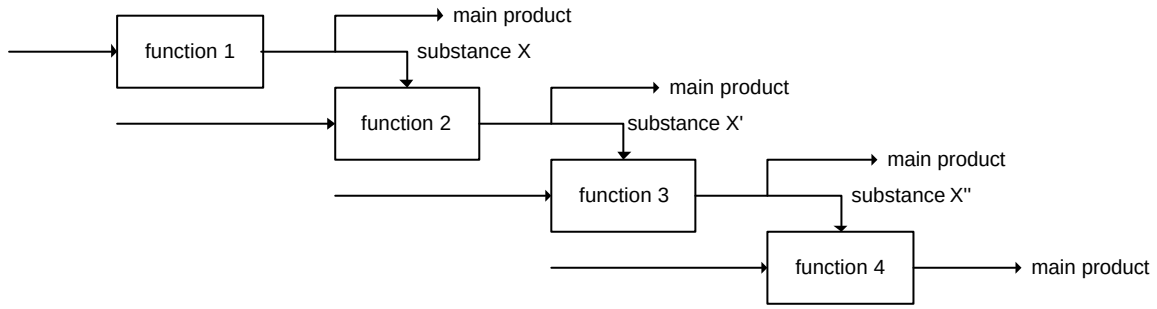


Figure B3: Cascade

Following function 1, X is used as a raw material for function 2, and then for function 3, and so on. The question, now, is what environmental interventions associated with the substance flow are to be assigned to each of the functions. This is a form of the allocation problem that is commonly encountered.

This terms of this problem have been described by Huppes & Schneider (1994) and Lindfors *et al*. (1995a). It has been clearly elaborated by Kortman *et al.* (1996), who distinguish between allocation to the upgrading step (from residual by-product to secondary material) and allocation of the interventions associated with primary extraction and production.

[1] Some literature sources mention NGOs as a separate stakeholder category. Although NGOs certainly play a specific role in the decision-making process, in the 'issue' phase that role can be considered embodied in the stakeholder 'consumer'.
[2] New in the sense of familiar, recognised applications.

The cascade problem boils down to the question of how the interventions associated with the processing of a particular substance flow are to be divided over the cascade of functions. In answering this question, criteria might be drawn up that relate to the degree of stability of the chain. in other words: to what extent are sales of the substance flow from function (j) guaranteed for the purpose of function (j+1)? This type of consideration accords with the basic principle of an *economic* supply chain.

These insights can readily be applied in classical LCAs (i.e. for product comparison) involving bulk materials like paper, plastics and metals. Particularly for chains involving aluminium, the method is very sensitive and therefore requires a broad support base when it comes to the allocation method employed. For this particular material the cut-off method has little credibility (Ven, 1996).

4.2 Lifestyle

The economic activities of society are geared towards satisfying perceived needs. There are several main groups of functions: transport, shelter, food, clothing, leisure, protection, etc. The list is certainly not exhaustive. A characteristic feature of function delivery is that it generally involves a combination of a physical function and fulfilment of a specific need.

> Example: the function 'shelter' can be described in terms of *protection against climate, disposal over living space, comfort, etc.* In addition, though, the location, neighbourhood, scenery, region and so on all contribute to the overall notion of 'shelter', or 'living function'.

The combination of physical and non-physical satisfaction of needs is played out at the level of the individual. At that level 'the function' is beyond the capacities of LCA. Matters might be different if clusters of characteristics could be identified in predictable combinations, in terms of 'lifestyle', for instance. The following lifestyles might then be distinguished:

- childless, working couples;
- the young, with their disco and 'house' culture;
- young families;
- senior citizens.

The idea would then be to assign a distinctive 'package' of products to each of these lifestyle clusters. The functional unit could then be defined as '1 year of life of the reference group'. Table B5 shows a selection of product packages that might be used for this purpose. These are all products that are situated within the economic supply chain.

Table B5: Examples of products associated with lifestyles

domain	example
food and diet	- confectionery, delicacies
	- drinks
	- meals
	- restaurants
clothing	- fashion items
shelter	- living space
	- furniture
	- household appliances
	- domestic help
leisure activities	- recreation
	- holiday(s)
	- clubbing
mobility	- car
	- other vehicles

It is no simple matter to incorporate 'affective value' in the functional unit or, in the case of a comparative LCA, to render the functional units 'equivalent' in this respect. At any rate, the 'affective function' cannot be elaborated in a similar manner to the physical function. When developing a policy support tool it is better not to mix the two worlds and it is therefore recommended to restrict LCA to the physical function, using it as a tool to elucidate the environmental effects of the physical functional unit. This would allow

for assessment of the overall environmental impact of a given lifestyle. Examples of combination functions in which this kind of analysis might play a part in government decision-making and in consumer information campaigns include:
- clothing (e.g. use of various fabrics and dyes);
- household management (e.g. appliances to replace manual activities);
- food and diet (e.g. substitution of animal proteins);
- leisure (e.g. different forms of leisure activity).

4.3 Infrastructure and transportation

Physical planning is traditionally a controversial issue. In our present context we use the term as covering both the planning process as such, encompassing the combination of housing, work and transportation functions, and the planning of subsequent project implementation. Many disciplines currently contribute to the decision-making process, and the question is: what is the added value of LCA?
This topic is similar to the 'lifestyle' issue, in the sense that here, too, there is an 'affective' element to how infrastructure is elaborated (e.g. urban planning and landscape architecture). In this case, though, the physical function is not entirely univalent. Besides housing and transport, other economic functions may also be involved in decision-making, such as employment. As an initial approach, the same procedure could be adopted as for 'lifestyle', i.e. a chain-type procedure for establishing the environmental impacts associated with delivery of a physical function (with the alternatives), with the results being used as a 'mirror' for assessing the other, non-physical functions.
Disregarding for the moment how the affective element is to be elaborated in decision-making, in broad terms it can be stated that the physical function is an issue for which 'transverse' elaboration is eminently suitable. At best, assessment extends to direct, local impacts (EIA). It is not common for LCA to be employed in such cases.

Examples of issues on which LCA might be able to shed new light include:
- analysis of regional transportation systems (rail, metro, bus, car, bicycle, etc.), including infrastructure requirements[1];
- local and regional energy supply (heat and power production and distribution, including new technologies such as fuel cells);
- urban water cycles (consequences of emission abatement measures);
- optimisation of building and civil engineering structures in the urban environment[2].

If LCA is used for assessing infrastructure projects, appropriate system definition is crucial. One possible solution might be sought in defining the function as narrowly as possible, i.e. as construction of the 'physical installation' and maintaining it for a certain length of time. Functions which in turn make use of the 'installation', for which it is thus a background process, can then be ignored.

4.4 Waste processing systems

Waste processing is the final link in the production chain. The product has served its purpose and must now be disposed of. Obviously, the environmental burden associated with the disposal process must be allocated to the product. The question is where waste processing ends and where the natural environment starts, as is illustrated by the example of landfill. What this in fact means is that the end of the supply chain is not precisely defined. The chain ends with a quantity of residual waste, which can in time give rise to environmental emissions. In current LCA practice thermal processing of waste (incineration, gasification, pyrolysis, digestion) is sometimes included in the chain. Several simple

[1] There are indications that the alleged 'environment-friendliness' of rail transport compared with alternative modes of transport is based on a disregard of the environmental impacts associated with maintenance and infrastructure.
[2] Studies point to an interesting trade-off between engineering features (insulation) and air conditioning requirements.

landfill models are also available. The conclusion is that although the supply chain can indeed be extended to include part of waste processing, there remains some quantity of ultimate waste with a potential environmental impact (Finnveden & Huppes, 1995; Udo de Haes & van Halen, 1997).

National and regional governments are interested in establishing the preferred mode of waste processing for a given quality of supply, which may consist of a single (waste) product, several categories of product or an aggregated waste flow (AOO, 1995). In the last case, it is not always relevant to allocate over the various input flows, provided a fixed input quality is assumed. In scenario studies in which the macro-composition of the waste flow is subject to variation, due allowance should be made for the subsequent influence on emissions[1]. This requires a dynamic model (Rijpkema, 1996). Besides this issue of multifunctionality on the input side, waste processing also frequently involves multifunctionality on the output side, for not only is the function waste processing delivered, so too are co-products (electrical power and heat in the case of incineration, fuels in other thermal processes). This second type of multifunctionality must always be addressed. Both these multifunctionality problems are briefly examined in Annex 2.

Given the fact that waste processing is responsible for a substantial proportion of the environmental impacts associated with a large number of consumer products, product policy (both government and industry) would benefit enormously from improvement of both allocation methods, i.e. multifunctionality on the input and output side. However, a certain measure of decision support is already feasible with current LCA methods and is therefore desirable. Analysis can be focused on product improvement, based on the principal trouble spots in waste processing and on a comparison of processing methods for a given product[2].

In summary, it can be concluded that there are workable models for including waste processing in LCAs. This is relevant for the majority of products. If there are temporal issues associated with interventions and their resultant impacts (in the case of landfill, for example), LCA will be a less appropriate tool, however. In addition, allocation methodology must be further refined, bearing in mind the broad range of processes that must be addressed. LCA certainly has a role to play in supporting waste policy (national and EU) and is an appropriate tool for that purpose. It could, in principle, be used for further prioritisation of waste processing policy options, for example.

5. Conclusions and recommendations

LCA is an instrument for assessing a given system: a product or a group of processes (a product being conceived of as a particular embodiment of a set of processes). The defining characteristic of such a system is that it addresses a particular demand in society; it fulfils a certain function. The system gains its coherence from the economic supply chain and an LCA accounts for every stage in the life cycle of the system. Actual assessment focuses on the system's impacts on the environment. The chain of cause and effect underlying these environmental impacts is complex, and the assessment parameters employed in LCA are therefore merely *indicators* that stand for *potential* impacts. Many of them are so-called 'stressors', i.e. indicators referring to the start of the causal chain.

Consequently, LCA is eminently suitable for addressing chain-oriented issues. The LCA methodology allows the impacts occurring at each link in a given supply chain to be aggregated, down to the level of relevant 'sub-activities'. The outcome is, of necessity, an *indication* of the potential environmental impact of the product in question.

LCA is less appropriate as a tool for assessing the environmental impacts associated with a specific locality (a specific ecological system). Neither is LCA the preferred instrument for evaluating systems in which the temporal aspect of the intervention-impact chain cannot be ignored. A third task for which LCA

[1] An example of an analysis topic might be: what is the influence of separate plastics waste collection (mixed plastics, or just PVC, say) on the environmental impact of a waste incinerator?
[2] An example here is the question whether it is better to incinerate hazardous waste in a dedicated facility (e.g. rotating furnace) or use it as a co-fuel in a cement kiln.

is inappropriate is assessment of systems in which the ‘function’ comprises a blend of physical and ‘affective’ aspects.

New applications of LCA are to be sought in the category of complex functions, such as:
- multifunctional systems (including cascades);
- collective functions (including infrastructure);
- clusters of functions (including lifestyles).

Appendix C: Partitioning economic inputs and outputs to product systems[1]

Erwin Lindeijer (IVAM Environmental Research) [2]

Gjalt Huppes (CML)

[1] This appendix, drafted in 1999 as an interim result of the project, has in some respects been superseded by the (simpler) main text. As it goes into more detail on some points, however, it has been included here as an appendix. While the terminology has been aligned as far as possible with that of the main Guide, this is not true of all the positions adopted here.

[2] Currently employed at TNO-Industrial Technology - Division of sustainable product development, Eindhoven.

1. Introduction: problem definition and the ISO partitioning procedure

According to ISO standard 14040 (1997E), allocation is the partitioning of the economic and environmental inputs and/or outputs of a process to the product system under study. The problem of partitioning arises from the fact that in LCA we wish to break down the complexity of the economic system to analyse just one function (a product or service). This implies a need to draw boundaries around the analysed system and deal with the inputs and outputs of that system in a consistent manner. For this reason, defining system boundaries and performing allocation are allied problems. Also relevant to a discussion of partitioning are certain basic modeling choices. These are discussed in relevant chapters of the present Guide. One key modeling choice that has been made is for change-oriented LCA. If a different choice had been made, this would have led to different considerations and different results.
Because of the close relationship between the definition of system boundaries and partitioning, we begin our discussion by reviewing the basic approaches to subdividing (economic) systems and presenting the ISO standard procedure for allocation, as this is the only standardised procedure currently available.

The multiple economic inputs and outputs and environmental interventions associated with any given product system are always based on one or more multifunctional processes within the system. It is around these processes that the discussion on partitioning revolves. There are three basic types of multifunctional processes that require partitioning (Figure C1): multi-output processes, multi-input processes and input-output processes which cross system boundaries.

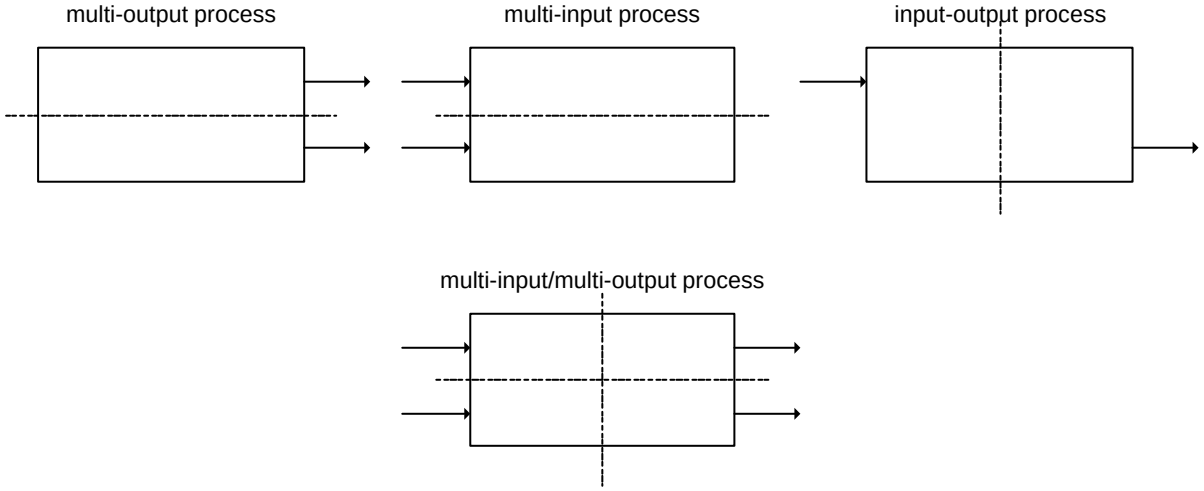


Figure C1: Basic types of multifunctional processes, and their combination. Functional flows to which all other flows are to be allocated are shown as arrows. Other flows have been omitted from the figure. System boundaries are depicted as dotted lines.

Although ISO 14041 (1998E) provides a basic stepwise procedure for addressing these processes which specifies preferred methods for dealing with economic system boundaries, it does not go into specific details. This allows for different interpretations and means that the procedure cannot be consistently and unambiguously applied. Nonetheless, it is an agreed framework to which we adhere in this Appendix, but which we shall interpret in two distinct ways. The text of the ISO procedure is reproduced below (the word ‘cannot’ in various phrases should be interpreted in the sense of ‘is not preferred for good reasons’).

ISO allocation procedure (ISO 14041, 1998E; Clause 6.5.3)

Step 1: Wherever possible, allocation should be avoided by:
- dividing the unit process to be allocated into two or more subprocesses and collecting the input and output data related to these subprocesses;
- expanding the product system to include the additional functions related to the co-products, taking into account the requirements of 5.3.2. *(Function, functional unit, alternatives and reference flows)*

Step 2: Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way which reflects the underlying physical relationships between them; i.e. they shall reflect the way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system. The resulting allocation will not necessarily be in proportion to any simple measurement such as the mass or molar flows of co-products.

Step 3: Where physical relationship alone cannot be established or used as the basis for allocation, the inputs should be allocated between the products and functions in a way which reflects other relationships between them. For example, input and output data might be allocated between co-products in proportion to the economic value of the products.

ISO 14041 also provides a number of general criteria and constraints on the partitioning of inputs and outputs that are of relevance to any further elaboration of partitioning methods. These principles are summarised below. Where adherence is obligatory for performing an LCA according to ISO standards, this is indicated in bold typeface. For purposes of consistency, some statements have been slightly paraphrased in line with the terminology employed in this guide.

1) The sum of the inputs and the sum of the outputs of the allocated subsystems **shall** equal the inputs and outputs of the unallocated system, respectively (100% rule).
2) Whenever several alternative allocation methods seem applicable, a sensitivity analysis **shall** be conducted to illustrate the consequences of the departure from the selected method.
3) There **shall** be uniform application of allocation methods to similar inputs and outputs of the system (*for open-loop allocation: recycled material entering the product system should be treated in the same manner as similar material leaving the system*).
4) When changes occur in the inherent properties of materials during subsequent uses, these changes **shall** be taken into account in the calculation.
5) When there are no changes in the inherent properties of the recycled material, use of that material displaces the use of virgin materials, and allocation is avoided by treating the product system as a closed loop.
6) In the case of open-loop recycling allocation there are several possible allocation parameters:
   - physical properties
   - economic value
   - the number of subsequent uses of the recycled material.

These can be considered as the ISO requirements for dealing with the partitioning of system inputs and outputs among product systems. In the next section we examine several specific approaches to partitioning, following the ISO allocation procedure and thereby explaining how system boundaries are to be drawn. Section 3 is devoted mainly to step 1.2 of the ISO procedure (expanding system boundaries, or substitution). Step 1.1 (division into subprocesses) should be dealt with as part of inventory modeling and is here mentioned only briefly. Section 4 discusses ISO step 2 (allocation based on physical relationships), pointing out the close relationship with step 1.1, reflected in the examples given by ISO to illustrate this step. Section 5 discusses economic allocation, the main example of step 3 of the ISO procedure. Allocation in proportion to mass, in certain applications a suitable proxy for allocation based on economic value, is also briefly discussed. In the final section of this appendix more precise criteria are elaborated and guidelines provided for applying each step of the ISO procedure, with the aim of reducing its ambiguity and achieving more consistent usage. In particular, two main interpretations of the ISO procedure are provided.

2. Basic approaches to allocation and the question of system boundaries

This section, based on a literature survey, discusses the principal partitioning methods currently in use for the three types of multifunctional process illustrated in Figure C1, indicating their respective ranges of application and their relationship to the steps of the ISO procedure and examining their implications for redefining system boundaries (see Section 1). From this discussion a number of basic approaches are distilled, which are discussed in subsequent sections.

MULTI-INPUT PROCESSES

The main type of multi-input process in which partitioning is required is waste disposal, e.g. landfilling, incineration or composting. This complex subject has been addressed in a comprehensive Dutch project (Udo de Haes & van Halen (eds), 1997) as well as in a number of specific methodological studies (e.g. Eggels & van der Ven, 1995; Finnveden, 1996c; Bez *et al.*, 1998; Nielsen & Hauschild, 1998).

In all these studies, close examination and modeling of the respective disposal routes allowed allocation to be avoided by division into subprocesses, or allocation to be based on physical relationships (e.g. chemical properties) for all the components of the input waste streams (step 2 of the ISO procedure). If the electrical output of an incineration plant is allocated to the various inputs according to their heating value, this can also be considered as application of ISO step 2. After allocation of the electrical output, there still remain input-output allocation problems, however: metallic outputs with an economic value, for example[1]. Dealing with the multi-input part of such processes is a highly technical, process-specific issue and the reader is referred to the literature. The input-output component will be considered in more detail below.

Transportation of packaged goods (the example of 7.3.1. in ISO TR 14049 (1998) for step 2 of the ISO 14041 procedure) is also a multi-input process, leading to a similar process-specific solution: partitioning the load-dependent part of the burdens depending on load weight or volume[2]. Again, the reader is referred to the literature (FhG *et al.*, 1991). In this case no input-output allocation problem remains, as the process is part of the main product life cycle.

Inputs are generally followed upstream to the ultimate raw material inputs from the environment system, or to their source in other product systems, as secondary materials. In our present context it is this issue of secondary materials input that is important, for these are to be partitioned in the same way as similar secondary materials leaving the system (see criterion 4 in Section 1, above). The present discussion of partitioning approaches already rejects out of hand any method not satisfying this criterion. In examining each of the allocation methods below, we shall specifically indicate how these inputs should be dealt with.

MULTI-OUTPUT PROCESSES

Many approaches have been proposed and discussed for partitioning multi-output processes (e.g. Huppes, 1993; Udo de Haes *et al.*, 1996; Heijungs, 1997a; Klöppfer, 1996, Frischknecht, 1998; Weidema *et al.*, 1999). The majority of these methods avoid allocation by system expansion (step 1.2 of ISO 14041) or allocate flows on the basis of non-physical relationships (ISO step 3). The only multi-output example found for ISO step 2 (in the 'Eco-labeling' report, Udo de Haes *et al.*, 1996) seems to be more exemplary of ISO step 1.1 (see Section 4). Here, our focus is on approaches according to ISO steps 1.2 and 3, especially those dealing with recycling.

*** Step 1.2 of ISO 14041 applied to multi-output processes**

[1] After the process has been subdivided up into discrete components per input, many input-output processes will remain. These will sometimes be multi-output processes (burning a hammer with a wooden handle, for example, producing iron and electricity), but at this detailed level of modeling the various outputs can be allocated to components of the input, with some simplifying assumptions. Consequently, input-output allocation should ultimately treat each component separately.

[2] In this example the process 'transport' can be subdivided into transport of commodities and of packaging, permitting independent variation of inputs within the constraints of maximum loading weight and volume. This means that step 1.1 of the ISO procedure is in fact used for analysing the system.

There are a variety of practical options for avoiding allocation by system expansion (step 1.2 of the ISO procedure).

Classical system expansion
Direct avoidance of allocation in the true spirit of ISO step 1.2 proceeds by expanding the system boundaries to include all the additional functions resulting from multifunctional processes in the system. As this involves introduction of a new, more broadly defined functional unit, however, it alters the very goal and scope of the LCA study. This is not generally an option for all multifunctional processes, for it implies an extension of the analysis to the whole world, as each expansion itself involves multifunctional processes. This approach may be feasible for major functional outputs only. To render 'equivalent' the systems being compared, to all alternatives not having the same additional function a system is then added that provides just this additional function. To avoid the multiple functional unit, the additional system is then subtracted from all alternatives. This subtraction of the additional function system can be interpreted as substitution. For example, the electrical power co-produced in waste incineration substitutes primary power production and can be subtracted from the multifunctional system to render it monofunctional.
A special case of system expansion involves extension to an entire product group in a given region, as has been done for European wood fibre products (Ekvall *et al.*, 1997), for example. In this case detailed models of subsystems are interlinked according to the annual material flows within the market in the region concerned. The same principle has also been adopted for a database on corrugated cardboard (FEFCO, 1997), although in this case there is less detailed analysis within the system. In all these cases, the need for allocation is restricted to the (extended) system boundaries. On the implicit assumption that the net material inflows and outflows are of only negligible (economic) value compared with the flows within the system, a cut-off is introduced. Multiple outputs of less important processes are dealt with through cut-off or quasi-closed-loop (sometimes termed semi-closed loop) procedures (see below), where flows actually going to other product systems are assumed to be used in the product system studied. Actual inputs are then treated as being substituted by the semi-closed loop flows. In these examples the goal and scope of the studies allowed for a 'rough and ready' cut-off, to be subsequently validated in a sensitivity analysis. It should also be stressed that in such cases it is no longer possible to analyse individual products within the product group of interest, unless allocation or substitution is performed at that level.

System expansion and substitution
The system expansion concept can also be applied to recycling. The reasoning often applied is that this is theoretically equivalent to performing a comparison between a function system (1) delivering two functions x and y and two other systems (function systems 2 and 3) each delivering only one of the functions (see Figure C2 below). By subtracting the single function system 3 from the multifunctional system 1, this combined system is reduced to a system performing just one function. This reasoning can be simplified by saying that the secondary output y of function system 1 replaces the primary production process PROD B of function system 3. This popular line of thinking forms the basis for the so-called 'substitution, or avoided burden method'. This method makes several implicit assumptions: on how material production for function system 3 is being replaced, on the absence of new market demand arising through substitution (i.e. substitution occurs only within existing markets) and on the feasibility of other multiple outputs within function system 3 being able to be dealt with in a similar fashion.
It is not always obvious which additional function should be subtracted. In everyday LCA practice a database is often used in which processes are already allocated, but this is merely a pragmatic choice with no general theoretical underpinning[1], as this begs the question of how the multifunctionality problem should be solved. Nevertheless, this is a generally accepted approach to substitution using monofunctional databases.

A special case of substitution occurs when the same type of material is substituted, but is not necessarily recovered in the same production system. An example is the reuse of polycarbonate from milk bottles in vehicle windscreens. According to ISO Technical Report 14049 (1998; clause 8.3.2) a so-called (semi-) 'closed-loop procedure' can be used to model this situation in which material quality remains the same, essentially using the avoided burden approach for the amount of material recovered

[1] Frischknecht (1998, p. 119) states that the avoided burden method may also be used in descriptive LCAs, but that the choice of alternative technology is then entirely arbitrary.

above the amount entering the system via the general market mix. ISO states as an additional requirement that the same primary and secondary production processes must be used at both the product and the (national) market level. However, this approach can also be used to model quality losses in a cascaded system. Care must be taken to closely analyse the absolute and relative amounts of material circulating in such a system. In fact, the product system studied is a subsystem of the total primary and secondary market system, resulting in a need to allocate (for instance) a net export of secondary material from the specific product system to the (possibly regional) market system, taking into account the attendant quality loss. The procedure for dealing with such subsystems is elaborated in Section 3, below.

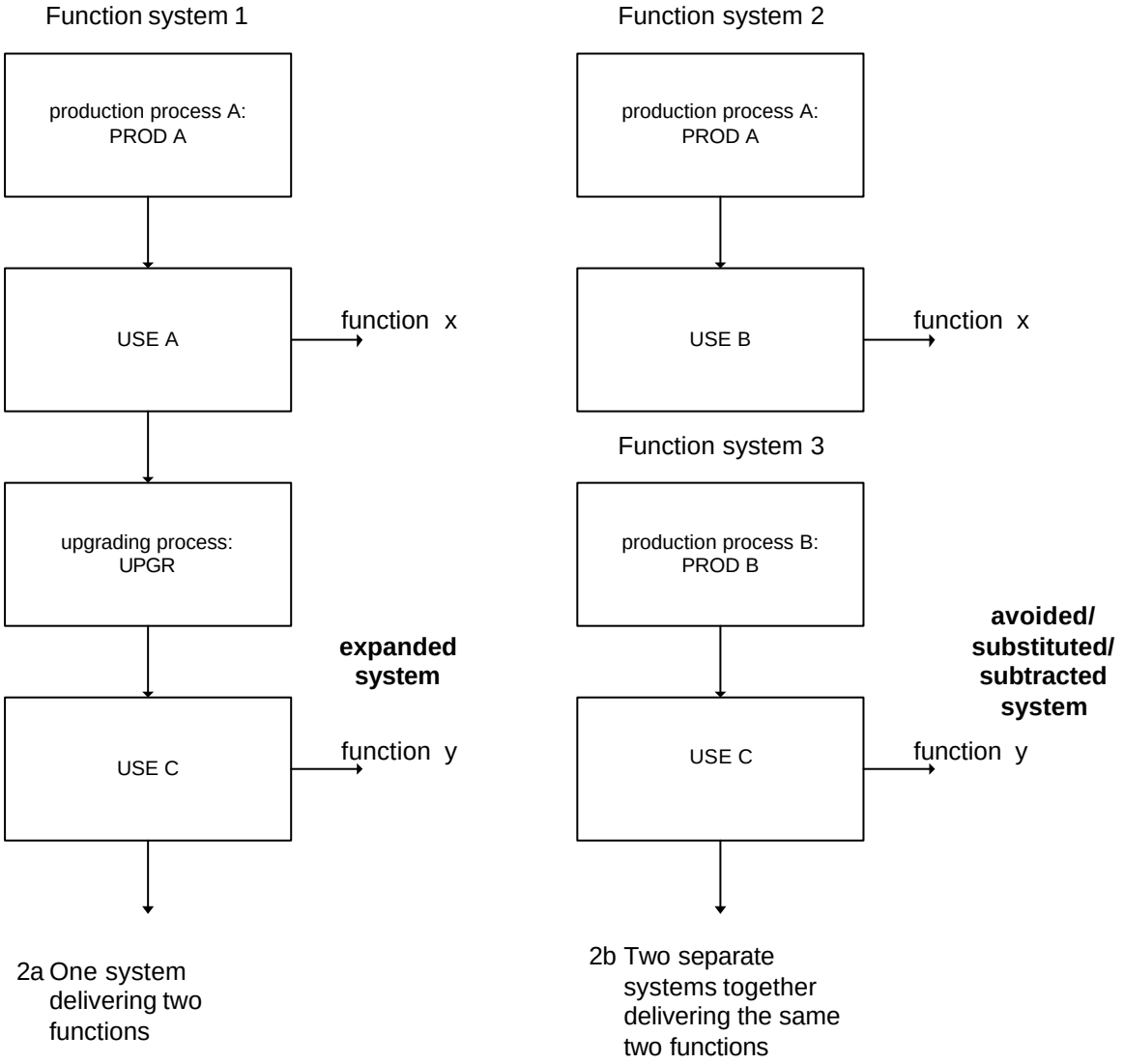


Figure C2: System expansion for recycling.

The '50–50 method' advocated as a default in the Nordic LCA guidelines (Lindfors *et al.*, 1995a) also implicitly assumes return of material of the same quality to the same system, or at least something equivalent. Fifty per cent of the production, waste disposal and upgrading processes is allocated to the product system under study according to the proportion of primary or secondary input. This approach is a compromise between rewarding use of secondary material and rewarding supply of recyclable material. The 100% rule is violated here as the 50% rule cannot be symmetrically applied to the connected systems, because primary production is also being halved, compared to the actual amount. This is a rough and ready method for dealing with systems as parts of larger market systems.

A related but more elaborate line of economic reasoning for applying the substitution concept is that the delivering system really will substitute other systems that might deliver the additional function, instead of theoretically comparing and substituting systems. This implies building a scenario for each future

substitution, based on market expectations. Seen in this way, in Figure C2 there is assumed to be an actual market shift in the future, with process UPGR replacing process PROD B, which hence can be subtracted, interpreted as substitution. This reasoning is more consistently in line with the change-oriented LCA framework. A procedure for dealing with such situations (as well as with other multifunctional processes) is given by Weidema *et al.* (1999), who explicitly state that it is valid for change-oriented LCAs only. These authors examine which processes will be substituted as a result of volume adjustments and which will not. Those processes characterised by volume adjustment (in economic terms, those with a high elasticity of supply) are the marginal processes. This approach is discussed below in Section 3.

Cascade approaches

Frequently there is a cascade of lower quality applications, as is the case when white paper is recycled into journals and journals into cardboard. In such cases where the quality of secondary material does not remain constant, the closed-loop procedure cannot be applied in the same way. Neither can the substitution method be directly applied if there are no processes substituted by the different material quality, as is the case, for example, in low-quality, secondary use of mixed plastics. Such low-grade plastics are simply not produced in a primary production process and hence there is no substituting process. Either the lower-quality material is incorporated in a general model analysing only an average material quality (as in the market-level, closed-loop model; see above) or the quality loss must be taken into account by reckoning with the quality loss occurring in each cycle (the cascade approach). The latter approach can be seen as a form of 'system expansion'.

Clause 8.3.3 of ISO TR 14049 (1998) gives an example of the cascading approach for kraft bleached paperboard being recycled to tissues, with no further recycling, and to other products, which may be further recycled. Accounting for quality loss may partly be achieved by taking into account the material losses occurring in each cycle[1]. However, the method applies an inappropriate mathematical formula (assuming infinite recycling, although paper fibre cycles are limited in number). Using the method without the simplified formula may be feasible, but is probably tedious and has not been encountered in the literature.

Other examples of the cascading approach are the so-called 'quality method' (Lindeijer, 1994 and Kortman *et al.*, 1996) and the 'estafette method' (Seijdel, 1994). The most elaborate method in this category is that of Schneider (1996), which analyses the complete cascade in full detail. In the quality method, the primary production is divided among all the cycles according to the relative quality loss per cycle, and the losses in volume (or mass) in each cycle, as waste. The waste disposed of in each cycle includes the discarded lower quality material. The upgrading process is allocated to the receiving system, by convention, although allocation based on a consistent choice of system boundaries is preferred. In the estafette method each cycle is considered to substitute a different primary material (according to its quality), effectively applying the substitution approach to each cycle. In this method, however, final waste disposal is also divided over the entire cascade according to the functional losses occurring at each step, requiring quantification of these relative losses. The upgrading process is necessary for delivering a certain quality output and is thus allocated to the delivering (or receiving) system. The method can be applied only to systems in which there is a primary process that can be appropriately substituted.

Note that there is no substitution of primary materials in either of the examples. Thus, recycling scores worse environmentally than if full substitution were assumed. All these methods require more or less detailed knowledge of (at least) the number of cycles and the quality and volume reduction occurring in each. All cascading approaches face the problem of major uncertainties regarding the quality arising in each cycle, making it generally hard to decide which burdens should be allocated to which material quality pool. In the TR 14049 example on bleached kraft paper and in the simplified quality method applied in (Kortman *et al.*, 1996) the lower-quality systems cannot even be distinguished from one another.

Applying these methods consistently in databases is therefore problematic. Even using them in such a way as to distinguish between individual product systems seems contrary to the fact that the whole cascade is considered and still requires allocation of the intermediate input-output process (see below).

Given these detailed problems, current cascading approaches are not considered adequate as basic solutions to the multifunctionality problem and may only be applied as non-default approaches in a sensitivity analysis.

[1] The associated waste disposal should also be taken into account. This is not mentioned in ISO TR 14049.

*** Step 3 of ISO 14041 applied to multi-output processes**

Recycling can be modeled as a multi-output process if the material retains a positive economic value throughout the system. There are then at least two functional outputs: the functional unit in the use phase and the discarded material resulting from that use[1]. On this line of reasoning, all multifunctional processes, including recycling, can be dealt with in the same consistent fashion. However, this requires that relative economic, i.e. market value be used as an allocation parameter, as physical parameters cannot be used for the functional output of the system. If economic value is taken as the decisive criterion for defining system boundaries, this is also the most obvious allocation parameter. (It is, of course, an important reason for the processes occurring at all[2].) This form of 'economic allocation' can be applied equally to input-output processes and multi-output processes. It will now be discussed briefly in relation to the former and explained in more detail in Section 5.

INPUT-OUTPUT PROCESSES

*** Step 1.2 of ISO 14041 applied to input-output processes**

Step 1.2 of the ISO procedure can be applied to input-output processes, particularly in the form of the avoided burden and marginal substitution methods. The system is thereby expanded to enable subtraction of the substituted production process. As mentioned earlier, the marginal substitution method requires scenario-building and, for fair comparison in comparative LCAs, also assumptions regarding the future development of markets. System expansion to encompass the full market of a coproduct (e.g. all electricity production) does not solve input-output allocation problems at the extended system boundaries; it somehow applies a cut-off method (see below) as a final resort, as do the other substitution approaches.

Step 3 of ISO 14041 applied to input-output processes

The position of the system boundary depends on the economic value of the flows leaving the processes of interest. Given a chain of processes in which discarded products from use process USE A (delivering function x) are upgraded in UPGR, which subsequently supplies use process USE C (delivering function y) five cases can be distinguished (Figure C3). If the value of the outflow of USE A is positive and paid for by UPGR, USE A is to be allocated at least partly to USE C, and UPGR entirely. The system boundary is then to be drawn in position 1, indicating that some fraction of USE A is to be regarded as a production process for the secondary material in USE C. If the discarded products from USE A have zero value, the system boundary (no. 2) is drawn between USE A and UPGR. If the discarded products have a negative value and the outflow of UPGR positive, the boundary (no. 3) is drawn in UPGR, which fulfils a waste management function with respect to function x. If after upgrading the value is zero, the boundary (no. 4) is drawn between UPGR and the downstream system, USE C. If the value is still negative even after upgrading, UPGR is to be allocated entirely to USE A, as is part of USE C, which fulfils a waste management function for it. System boundary no. 5 then pertains.

Now, how can ISO step 3 be applied? This is necessary only for system boundaries 1, 3 and 5. The simplest solution then is to treat 1 as 2, and 3 as 4. In the first case the upgrading from negative (or zero) to positive economic value is considered part of the downstream system, USE C. This is called the 'cut-off method' for allocation and can be considered a rough estimate for dealing with input-output processes at the system boundaries (i.e. when no reliable data is available on the economic value prior to upgrading). In the 1992 CML guide (Heijungs *et al.*, 1992) this approach was proposed as a simple, but unsatisfactory, general method for dealing with allocation.

The commonest situation in recycling is one in which a negatively valued discarded product, such as a waste (disposal of which must be paid for) is upgraded to a positively valued product. Here both the

[1] It is arguable whether the functional use is actually an economic output: no one pays consumers for consuming products the way producers are paid for their economic output. Not regarding the functional output as an economic process would change the situation from multi-output allocation at the use process stage to input-output allocation at the upgrading process stage (see below). Which line of reasoning is followed is crucial. In section 5 the starting point is that the functional use is an economic output.

[2] In some cases the market price may not be the most appropriate allocation parameter, especially if the market is not fully developed. This is often the case for new recycling initiatives and an alternative parameter must then as a proxy.

costs and the environmental burdens associated with upgrading are borne by the receiving as well the delivering system. The process where the value switches from negative to positive is considered an input-output process that traverses the system boundaries. The environmental burdens of that process are then to be allocated partly to the delivering system and partly to the receiving system. In the 50–50 method discussed above (Lindfors *et al.*, 1995a) 50% of the upgrading process is allocated to the delivering system and 50% to the receiving system. A more sophisticated method, however, is to allocate according to the shares in total proceeds of the process. Similar to the situation with boundary 1 is that in which the economic value of the material in the discarded product remains negative even after upgrading. This implies that the receiving system is providing the delivering system with a form of waste management service. Some of the burdens of the receiving process should therefore be partitioned to the delivering system, in as far as these burdens have not been separated off under step 1.1 of the ISO procedure. The upgrading process is then part of the delivering system.

Economic allocation

A generally applicable method for all system boundary situations concerned is economic allocation, whereby the various functional flows are allocated according to their share in the total proceeds of the process. This method is also applicable to the other types of multifunctional processes.

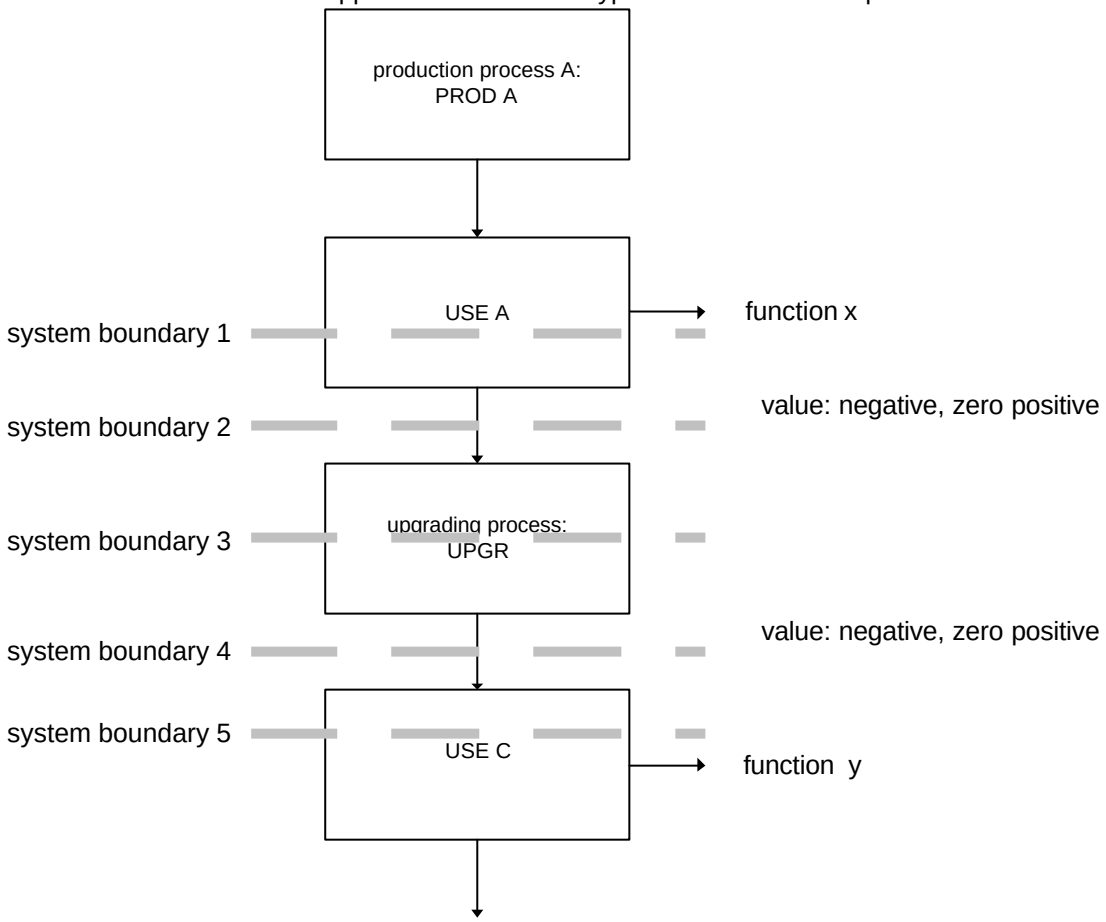


Figure C3: System boundaries and allocation in recycling (UPGR: upgrading process from material USE A to USE C)

Conclusion

The problem of partitioning is a complex one and many solutions have been proposed. We have endeavoured to structure the discussion by distinguishing between various types of process, related to different system boundary problems, and following the steps of the ISO allocation procedure, in the context of change-oriented LCAs.

As has been illustrated, the partitioning approach is closely linked to the delineation of boundaries around the product system(s), and most approaches apply specific criteria for doing so. The goal and scope of the study are also important here, as system expansion means a redefinition of the functional unit. Two basic approaches can be distinguished:

1) avoiding allocation by expanding the system boundaries (ISO step 1.2, by generalising the product system(s) under study to be of a certain average material quality and applying a closed-loop allocation procedure, or by analysing the system marginally substituted by the recycled material and subtracting it)
2) allocation at the system boundaries at a process level (ISO step 3), here based on economic considerations, i.e. market value or imputed market value.

The marginal substitution method always draws the system boundaries around processes, while the economic allocation method generally draws them within processes. Neither method takes cascading effects into account. The two approaches are discussed more extensively in Sections 3 and 5, respectively. Allocation on the basis of physical relationships between outputs (ISO step 2) is deemed feasible only if the relative magnitude of outputs can be changed, i.e. if it is a combined process, rather than a joint process in which relative outputs cannot be varied. This situation is examined in Section 4.

3. Avoiding allocation (step 1.2 of ISO 14041)

RULES FOR SYSTEM EXPANSION

If sensitivity analysis shows that flows across system boundaries and stocks are negligible, cut-offs may be introduced. If this is not the case these flows must be taken into consideration, however. The simplest mode of system expansion is then to use a broader functional unit, i.e. expand the system being assessed. It should be checked whether substitution leads to serious nested-system boundary problems, as the systems added in expansion will themselves usually be multifunctional systems. For example, electricity co-produces heat and, more generally, most well-developed processes are multifunctional. In most cases, therefore, it is to be expected that allocation by step 3 should be applied. If substitution is possible, certain rules apply.

RULES FOR SUBSTITUTION

System expansion can be applied at a single product system level by the 'substitution, or avoided burden method'. The material substituted should have the same quality as the recovered material (the burdens arising in achieving that quality, e.g. in an upgrading process, should therefore be included in the system delivering the recovered material). This approach should only be applied in cases where it is entirely clear which material is (to be) substituted. As this will never be clear without a market study, this approach is very coarse and a sensitivity analysis should therefore be performed to indicate the difference in results if different substitution is assumed or some allocation method is applied. (Sensitivity analysis should of course be undertaken, as appropriate throughout every LCA.) Furthermore, when PROD B in Figure C2 (situation 2b) is avoided in a functional unit (say 1 kg) of PROD A, only a fraction of the market for PROD B is replaced by PROD A. This fraction should then be incorporated in the modeling of PROD B. When comparing a system with incineration to a similar system with materials recycling, for example, it is not only the substitution of materials that needs to be addressed but also the reduction of electrical output due to less incineration, leading to additional electricity production elsewhere. Caution is thus in order when using data on unit processes from databases in which the substitution method has been applied, as the process adjustments on the input side may not have been calculated according to product-specific changes in outputs. Appropriate calculations should then be made to correct these adjustments throughout the database, to ensure that system imprecision does not lead to major fuzziness of the LCA results. In fact, we do not recommend the avoided burden method as a baseline approach, because of its indeterminate implications for the consistency of system modeling.

Simplifying the system by adopting a (semi-) 'closed-loop procedure' is equivalent to an avoided burden method in which the material quality remains the same. As substitution is simplified here and no marginal impacts are to be anticipated, this is a valid approach when the inherent properties remain the same, in accordance with ISO 14041 (1998E). If material quality is reduced relative to average material quality in the marketplace, allocation should be on the basis of this quality difference. One way to correct for lower quality is to subtract only a fraction of the primary material substituted, according to the ratio between the value of the primary material and the degraded secondary material.

Weidema *et al.* (1999) have outlined a procedure for selecting which processes are to be deemed marginal processes, i.e. which processes will be affected by a particular change and therefore substituted in the market. As the figure explaining this 'symmetrical substitution' procedure could not be reproduced here, a text summary is provided. See also the example by Weidema in the main text of Part 2b.

The procedure consists of five steps and seeks essentially to answer two questions:

1) What is the situation in which the studied change in demand occurs?
2) In this situation, which specific technology is affected by the change?

which are addressed, respectively, in steps a to c and d and e:

a) What is the time horizon of the study? This guide deals with long-term changes only[1].

[1] Short-term marginal effects may be identified using a similar decision tree as the one presented here. The difference between the two diagrams would be that instead of increases and decreases in capacity, the short-term diagram would show increases and decreases in capacity use, within existing capacity.

b) Does the change affect specific processes only or an entire market? If only specific processes: these are the marginal processes. If a market is affected, proceed to step c.
c) What is the volume trend in the affected market: upward or downward[1], that is are only old installations discarded or are also new installations installed? List possible technologies and make a choice.
d) Is production capacity anticipated to expand or shrink? If neither is the case, this technology is not a marginal technology; return to the list. If yes, proceed to step e.
e) Is this technology the most preferred (if being installed) or the least preferred (if being decommissioned)[2]? If it is neither, this is not a marginal technology; return to the top of the list. If yes, this is the marginal technology.

Although developed for selecting marginal technologies for general application in comparative LCA studies, this procedure can also be used for selecting the processes to be substituted in system expansion (Weidema, 1998a), as for the case of recycling. Asked for an example of application of this procedure to recycling, Weidema provided the following illustration:

EXAMPLE: STONE DEMOLITION WASTE WORKED INTO CONCRETE FILLER

The main output of the primary process is the service 'demolition'. The waste flow (or co-product; it does not matter, as will be seen from the following) is 'stone'. This then undergoes one or more recycling processes ('transport, crushing, washing, sorting'), before being fed as raw materials into a secondary process: 'concrete filler production'. How, now, should allocation of this recycling process proceed? This can be decided by asking one simple question.
If it is the secondary product that is the focus of the study, the question is 'Does an increase in demand for the secondary product ('concrete filler') lead to an equivalent increase in recycling?' If the answer is yes, recycling is the marginal production process for the secondary-process raw material in question, and the recycling process should be 'allocated' to, i.e. included in the product system of, the secondary product. If the answer is no, recycling is determined by external forces (from the perspective of the secondary product system), e.g. by legislation or because recycling is cheaper than waste treatment, and the recycling process should be excluded from the secondary system (and 'allocated' to the primary process).
If it is the primary product that is under study, a parallel question is asked: 'Does an increase in demand for the primary product (i.e. 'demolition') lead to an equivalent increase in recycling?' If the answer is yes, recycling is the marginal waste-handling process for the primary system, and is to be included there. If the answer is no, recycling is determined by external forces (e.g. demand for recycled products) and is not *to be included in the primary product system.*
As can be seen, these two questions are entirely complementary and lead to the same division between the two systems. Since we typically study only one of the two systems (the reason why an 'allocation problem' arose in the first place) we can amalgamate the two questions into one:
'Does an increase in demand for the product under study lead to a corresponding increase in the material turnover of the recycling process?' If yes, include the recycling process, if not exclude it. It should be noted that this procedure is independent of the price of the waste, i.e. it works whether the waste stone has a negative or positive economic value. It should also be noted that the word 'increase' can be replaced by 'decrease' throughout, with no change of result as long as we are dealing with small (marginal) changes.

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[1] To be precise, the option 'upward' is relevant only if market volume is declining *faster* than the decline resulting from regular, planned phase-out of capital goods. Consequently, the option 'downward' is also valid when market volume is decreasing *slower* than the regular capital replacement rate.

[2] The preference implied here relates to the expected long-term production costs taking into account all externalities relevant for the one who decides about the capacity adjustment. See the text on step e) for further elaboration of this point.

This example addresses upgrading processes only. According to the general reasoning underlying system expansion (Weidema, 1998a), if the delivering system is to include the upgrading process because an increase in its output fosters recycling, it should also be the one to which the avoided burdens of the substituted system should be allocated[1]. The same should also apply on the input side: if increased demand for secondary material increases output thereof, the avoidance of primary material should be accounted for (as has generally already been done in systems modeling). Finally, in a change-oriented LCA in which a steady-state model is used to compare 'business-as-usual' (e.g. building roads on sand) with a new situation (e.g. using demolition waste), market trends, both upward and downward, should be included in both cases.
In principle, the above procedure can also be adopted for non-marginal approaches (Weidema, 1998a). The modeling of the system should then be adjusted appropriately, however. For long-term non-marginal changes this means that future scenarios and technologies should be applied.

DRAWBACKS OF THE SUBSTITUTION APPROACH

Although this approach is the first preference in ISO 14041, several drawbacks should be mentioned:

- Classical system expansion can only be applied if the functional unit (and possibly goal) of the study is changed.
- In the case of marginal substitution, it is necessary to compare the situation with marginal (future) adjustments with the future situation without such adjustments. However, it is unclear how large an error will be introduced by comparing two future market scenarios calculated using current databases, in which the avoided burden method is generally applied.
- In the case of recycling, no operational procedure has yet been developed for selecting marginal technologies. What if supply of a secondary material follows demand, but not to an 'equivalent' (i.e. equal) extent? As substitution is then only partial, should only part of the process be subtracted? If so, it will be difficult to determine the appropriate extent.
- In the real world, supply and demand are to some degree elastic (i.e. volume varies in response to price), with complete elasticity or inelasticity extremely unusual. Both the general substitution method and Weidema's symmetric substitution method address these exceptions, rather than the general case.
- In subtracting a substituted alternative, there is a fair chance that the subtracted system will be a multifunctional one itself, leading to nested loops of subtraction. Arbitrariness may be introduced if a cut-off approach is applied after the first nesting.
- In the case of emergent markets, substitution may be ambiguous. It can be argued that demand for a new product made from recovered material will emerge regardless, and that in supplying for that demand equivalent primary production is thus being substituted, so that the first use of the material should be rewarded. It can also be argued, however, that such secondary demand will not necessarily emerge, in which case recycling should not be rewarded, as consumption of the product made from the recovered material becomes essentially a waste disposal phase, to be allocated to the first use of the material. The merits of the two positions are hard to judge in the abstract. However, this problem is relevant for all approaches for dealing with partitioning new material applications at system boundaries.

[1] If other co-products arise in the substituted flow diagram, the same procedure should be followed. If gravel extraction is being substituted by production of concrete debris, the question is whether the co-production of sand is also affected by substitution. If so, it should be included; if not, not. A sensitivity analysis should be performed to ascertain how far this procedure should be followed, but more than three levels will be rare.

4. Allocation according to physical relationships (step 2 of ISO 14041)

To indicate how step 2 of the allocation procedure in ISO 14041 (1998E) is to be elaborated, let us consider some of the examples provided in ISO TR 14049 (1998). Partitioning according to physical relationships among outputs (products) is exemplified in ISO TR 14049 by the combined transportation of packaging and goods, where only the packaging is analysed. Here purely physical parameters are used to allocate the energy consumed between goods and packaging: mass and volume. In this case the ratio between the various environmental parameters ({Di} in ISO TR 14049, clause 7.3.3) can be kept constant while varying the relative weight or volume of the outputs. The possibility of varying the relative outputs while keeping {Di} constant is stated as the sole argument for performing step 2 of the ISO 14041 procedure.

Example 7.3.2 of ISO TR 14049 (production of cream and low-fat milk from crude milk) is a different type of situation. It is presented as a hybrid between allocation based upon physical relationships and upon economic value. It is argued that as high-fat cream is economically more significant (per kg), fat content might be the appropriate basis for allocation rather than mass. In the end, however, it is indeed economics that supplies the basic rationale for partitioning, fat content being an only part-valid indicator for economic value per kg. As with any economic process, the total price of its outputs is its *raison d'être* and the economic proceeds per kg of the different outputs is therefore the most reasonable figure for allocating the outputs. Of course, mass or any other physical parameter can also be used as an indicator for this relative economic value (being stabler than price, for instance, or easier to determine), but only if its value as an indicator has already been proven.

On this reasoning, step 2 of ISO 14041 is no longer to be considered a separate step in the procedure for partitioning; in fact, the relative added economic value of the respective outputs is the most basic allocation parameter available[1]. The next section therefore focuses on economic allocation, the most satisfactory allocation method available.

[1] The actual need for the various outputs in society is of course even more basic, but can rarely if ever be quantified. Only if the market value of proceeds deviates markedly from the value that would have arisen in a well-developed market should a more suitable price level be sought (as when a novel process has not yet been upscaled, so that development costs still weigh heavily).

5. Economic allocation (step 3 of ISO 14041)

5.1 Introduction

The situation examined here is that arrived at in the final step of the ISO allocation procedure. Where possible, allocation has already been avoided (although several remarks will be made on the topic); substitution has been postponed as an approach that is not systematically applicable; and physical allocation has been performed, as in some cases of combined waste treatment. Generally speaking, the core system that results will still be multifunctional. The aim of this final step, then, is to model the economic value of the inputs and outputs of this multifunctional system, so that these values can be used as an allocation key. To this end market prices are generally employed, these reflecting the basic fact that in a market economy production processes are ultimately driven by the proceeds they command.

In principle, this allocation method is quite straightforward, employing as it does the kind of cost allocation and cost analysis that is widely applied in business economics. Problems may arise, though, (1) when markets (and therefore prices) are lacking, (2) when prices are 'distorted', as with imperfectly functioning markets (e.g. monopolies), or (3) owing to government intervention. Examples of the latter include certain forms of subsidy and compulsory reuse, and processes that are subject to permits. These issues are elaborated in greater detail in Part 2b of this Guide[1]. Below, in Section 5.2, we briefly explore two examples, on a provisional, qualitative basis: use of housing demolition waste and incinerator floor ashes in road building, the first in a situation of market equilibrium and the second with government induced market failure.

Although our approach to economic allocation is based on traditional cost analysis, as employed by neo-classical economics, we deviate in several important respects.It is not the place here to embark on a general critique of neo-classical economics. It may be noted, though, that LCA represents one way of dealing with market imperfections usually treated otherwise by economists of the neo-classical school. In this sense, the very exercise of LCA is a response to the failure of markets to properly account for environmental effects.

First, however, we provide an outline of the basic principles of economic allocation.

Before the allocation problem is tackled, the system under study must be clearly defined, already based to some extent on price-related criteria (see Figure C4). If a physical flow is unpriced or has a zero or negative price, it should be allocated to the functional flow or flows. This, at least, is the case for all environmental interventions (flows 1 in Figure C4). With waste to be processed, the direction of payment determines whether the flow is to be allocated, or allocated *to*. If it is an inflow to process *X*, the waste is a functional flow and constitutes one of the flows to which non-functional flows should be allocated (flows 2 in Figure C4). If it is an outflow from process *X*, the waste is a non-functional flow and constitutes one of the flows to be allocated to the functional flows (flows 3 in Figure C4).
In the case of products, i.e. flows with a positive economic value, the direction of payment again determines whether the flow is to be allocated, or allocated *to*. If it is an inflow to process *X* (flows 4 in Figure C4), the product is a non-functional flow and constitutes one of the flows to be allocated to the functional flows. If it is an outflow from process *X* (flows 5 in Figure C4), the product is a functional flow to which non-functional flows are to be allocated.
Economic allocation can thus be based on the following rule of thumb. Environmental interventions, as well as all those flows for which process *X* pays other processes, are allocated to the functional flows, while flows for which process *X* is paid are allocated *to* (*cf.* the grey arrows in Figure C4).

[1] Although some of these issues might be covered by more complex types of economic modeling, such models are beyond the practical scope of LCA and are therefore not further considered here.

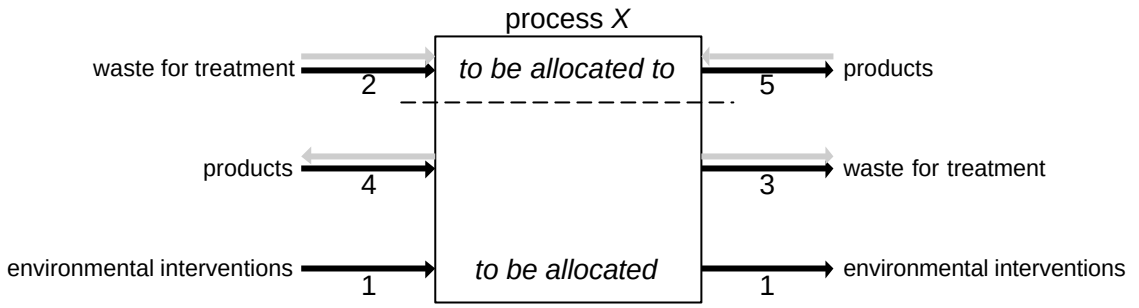


Figure C4: Flows to be allocated to functional flows and flows to be allocated *to* (black arrows represent physical flows, grey arrows monetary payments).

As the most general case we usually think of prices in market terms, i.e. as deriving ultimately from consumer preferences. Some goods and services may be government-provided, however, with no markets involved, as in the case of road building or education. It is then not final consumer demand in the marketplace that drives the economic chain, but public demand, driven by a politically guided decision-making process. As long as the products in question are purchased in the market, though, their price can be derived in much the same way as for consumer goods, for public demand is similarly reflected in the cost of supply, usually equal to the budget expenditure involved. In economic terms, the derived utility is at least as high as the utility forfeited by expenditure on the product in question (for otherwise something else would have been purchased). Thus, in the sphere of collective expenditures too, economic (i.e. market) value or budget expenditure can be taken as the basis for allocation.

5.2 Examples of economic allocation

By way of illustration, then, let us consider two examples.

CASE 1: USE OF HOUSING DEMOLITION WASTE IN ROAD BUILDING

After functional use, a house is 'discarded'. As a subsequent process the discarded house is demolished, with several inputs required and several flows resulting. One of these flows (pulverised bricks, say) can be used in road building, providing a stylised example that stands for the more general case of recycling. By varying the prices of the flows concerned, several system definitions and solutions to allocation can be illustrated.

To arrive at a system definition, it must be clear what question is to be answered and hence what functional unit is being analysed. One might, for example, wish to assess a specific technology for processing the waste flow in question and using it in a new application. In that case a difference analysis can be applied to the two systems together, with both functions represented in the functional unit. The alternative with which the comparison is to be made is a different mode of processing the demolition waste flow, e.g. without a functional application, and a different mode of supplying the input for road building, e.g. using primary materials. This analysis for technology choice is a legitimate one. It does not require allocation between dwellings and roads and in that sense is complete when the system boundaries have been set in the difference analysis. Allocation may still be necessary, e.g. for the multiple processes involved in manufacturing the plant for processing discarded bricks.

Alternatively, one might wish to compare different housing systems having the same functional unit, with one of the alternatives involving the three processes described in the example in Figure C5 below. Some form of allocation is therefore required here.

Four situations (a to d) are described here with respect to the values of the respective flows (see Figure C5). In variant a, the value of the discarded building is positive. The demolition processor pays the owner of the discarded house for being allowed to demolish it, thus paying for his 'secondary raw materials'. The building has then had two functions: housing, as the functional unit investigated, and materials provision. (With other products such as ships, this is quite often the case.) Part of housing construction

is then allocated to the function of materials provision. All the processes required for the functioning of the building solely as a dwelling, such as heating and cleaning, are there for that function only and are therefore not allocated to the discarded house. If the building materials (bricks, isolation material, window frames etc.) have been applied in the building in such a way that they can be separated only by being dismantled together, as may be reasonably assumed, the whole building demolition process is to be allocated partly to the housing function and partly to the materials-providing function. Then it is not some fraction of brick production that should be allocated to the brick waste, but also a fraction of total construction and maintenance, as being connected to the use process of the house. The resource use and emissions of the demolition process itself, and its other economic inputs and outputs, should all be allocated to the secondary uses. This example covers all cases involving two products from the same process (i.e. it is also valid for multi-output and multi-input processes).
Formula: Allocation to be based on the share of each function in total proceeds. Total proceeds are the sum of the two functions, the value of housing and the value of the discarded building.

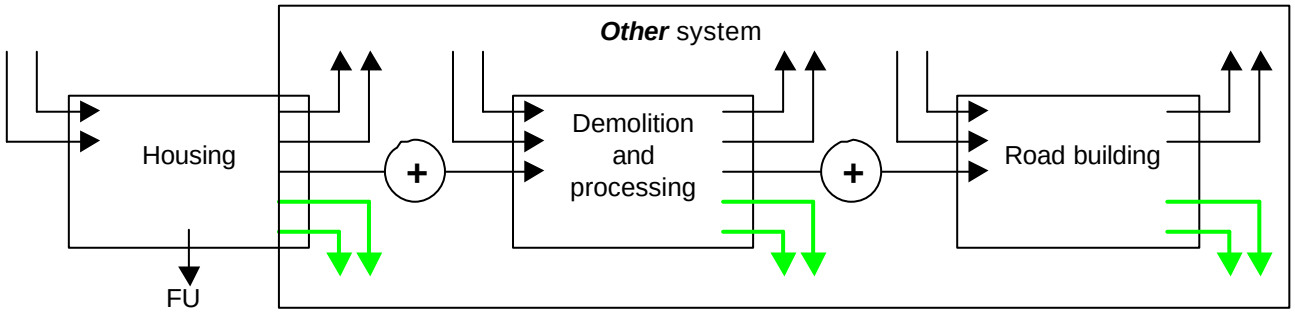
Variant b treats the special case in which the discarded house has zero value. In environmental terms, the input of the discarded house to the demolition process is then also 'for free'. The processing of the discarded product makes not contribution to the environmental interventions of the housing system. When comparing this to an alternative, such as dedicated waste processing without further applications, there is a clear advantage. As the boundary lies exactly between two processes, no allocation is needed. In some cases in which a discarded product has a low negative value, transport to the processor may be the first upgrading step to make the value (near) zero. The system boundary should then be defined so as to include this transport.
Formula: No allocation is required, as there is no process at the system boundary.

In variant c, disposing of the discarded house cannot be paid for from the proceeds of processing it, as in variant a, and the owner will consequently have to pay the demolition firm. Besides waste management, demolition, (including processes like sorting) will then also provide another function: one or more secondary products. In this case no fraction of building construction or maintenance is allocated to these secondary products.
Formula: Allocation to be based on the share of each function in total proceeds. Proceeds are from waste processing and from sales of secondary products.

In variant d, the flow used in road building still has a negative value, implying that its functional contribution to the road is negative, the road application essentially being a form of waste processing to be allocated to the functional use of having the house. We assume there are no other flows with a positive value (which would be equivalent to variant c). The system boundaries of the housing function then include the full process of demolition plus processing, as well as part of the next process, road building. The road-builder is then paid to accept the demolition waste and thus has waste processing as a co-product. *Formula:* Allocation to be based on the share of each function in total proceeds. The value of the road produced is measured in terms of its price, the value of waste processing in terms of payments to the road builder.

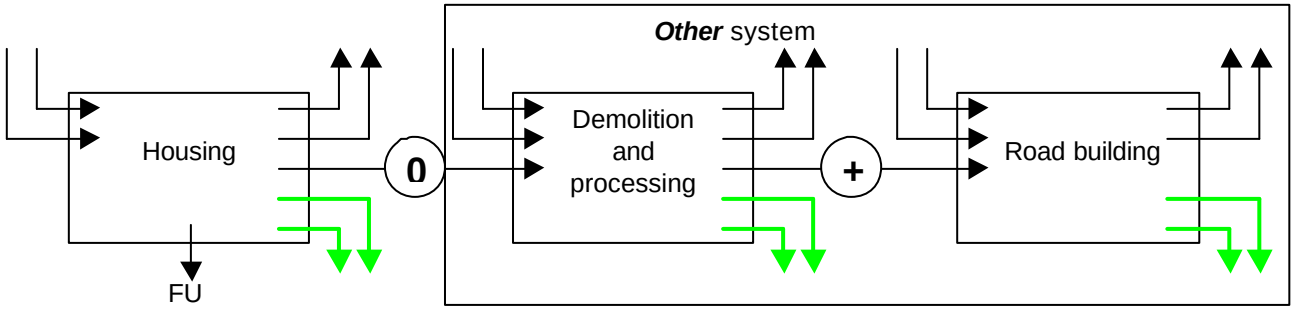
How to further interpret the outcome of this variant d? If the cost of using demolition waste in a road exceeds that of using primary materials and the environmental consequences are also greater, taking into account 'all' the environmental effects allocated to the road, the reasonable conclusion would be that promoting this type of secondary use by regulatory means would be erroneous. If the costs are higher and the environmental effects lower (or vice versa), a trade-off has to be made between costs and environmental effects.

Variant a: value of waste flow from building is **positive** and hence even more positive after processing and in application in road building.

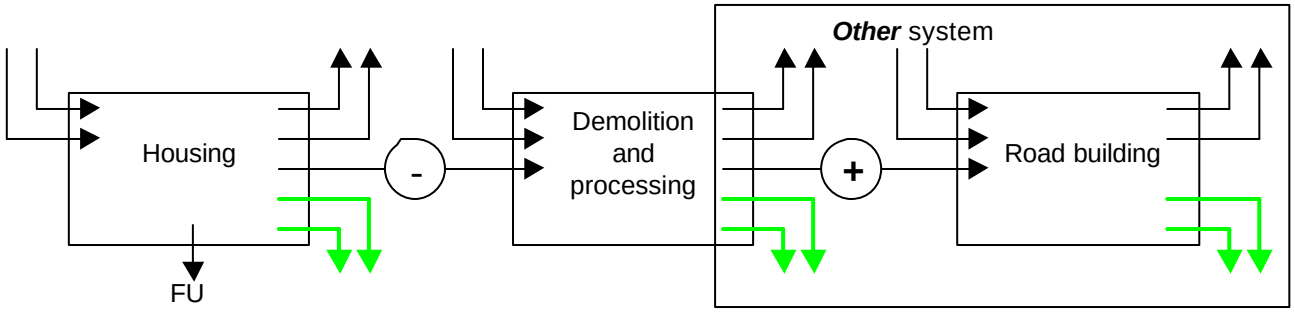


(FU = functional unit)

Variant b value of waste flow from building is **zero**, in road building **positive** and hence after processing also positive.



Variant c: value of waste flow from building is **negative**, in road building **positive** and hence after processing also positive.



Variant d: value of waste flow from building is **negative**, and after processing **negative**. Processing is a fully fledged part of waste management. Road building has the waste management function as a co-product.

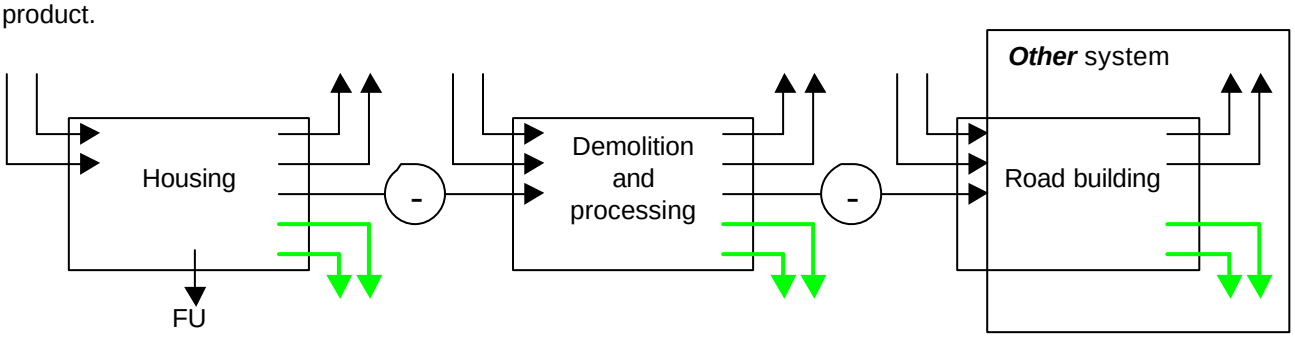


Figure C5: System boundaries and allocation: 'discarded house demolished and processed to road-building materials'

CASE 2: USE OF WASTE INCINERATOR FLOOR ASHES IN ROAD BUILDING

In the Netherlands floor ash from municipal waste incineration plant is used as a road building material. Its function is similar to sand, but the resultant road quality is somewhat higher. Because of the toxic substances in the ash, especially heavy metals, certain measures must be taken in road construction to isolate these substances from the environment. Also, after its useful life, the road must be demolished, with the incorporated ash having to be treated as toxic waste.

In this case there are two quite different questions on which LCA may provide decision support:

a. a comparison of different technologies for a particular process, e.g. waste treatment of incinerator ash, some of which yield co-products
b. a comparison of different alternatives for a particular function system, e.g. milk packaging incineration (as part of 'milk drinking') or road foundation construction, as part of a transport system.

a. Technology comparison

The first type of question is concerned with how a form of waste treatment producing a positive economic value compares with other forms of treatment, e.g. storing the ash in a controlled landfill and using sand, or gravel or other waste flows as a road building material. The functional unit is a combined one, comprising processing of a given waste flow and disposing over a given section of road for a certain period of time. This question requires no allocation between the waste-producing function(s) and road foundation construction. The analysis can be set up as a difference analysis between the two (or three or more) alternatives to be compared. Elements common to all of them can be omitted from the analysis. For each alternative, the same set of functions is delivered, waste management for the function of milk drinking and road construction for the function of transport. Although this procedure has similarities with substitution, here nothing is 'avoided': each alternative stands on its own.

The key question here is how the system boundaries are to be defined for the different ash-processing technologies. On the incinerator side, there is only one technical alternative, leading to the floor ashes from combustion. On the road side, there are different alternatives to be compared. In the difference analysis, common elements may be omitted from the analysis. Let us assume for simplicity that road quality, in terms of maintenance requirements and lifetime, is the same in all alternatives. The only additional item to be included in the road-building alternative using incinerator ash is then the environmental isolation of the ashes in road construction. After its useful life the road is 'discarded', at which stage the ash-related waste management activities must also be included in the analysis. The kind of waste management then required is a matter of regulatory technical specification. Let us assume that second-time use is unattractive, as the ash has become mixed with other materials and isolation is becoming increasingly difficult. Final disposal of the ash-containing waste must then be included in the analysis. Let us assume that use of primary sand is another alternative, using sand available nearby. When 'discarding' the road, the sand would not constitute a waste material but could subsequently be used as a stable filling material. Let us assume that the value as a filling material from discarded road is zero, so there are no further processes to take into account. By comparing the outcomes of the different alternatives, an indication can be obtained of which technology is environmental superior. Although allocation may still have to be applied to upstream processes, e.g. those producing plant and equipment and energy for transport, in all other respects it will have been avoided, by formulating the question in a specific manner in which values and market prices play a role in only in setting system boundaries. no role.

The Dutch government has made it more or less obligatory for incinerator floor ash to be used in road construction. Is this indeed the best processing option? This question can be answered only by comparing this option with the relevant alternatives, in the manner indicated. As LCA treats environmental effects now and in the future consistently (there is no 'discounting' of environmental effects) the outcome of the analysis might very well be that it would have been better to store the ash immediately, especially as more effective isolation measures could then have been taken. Toxic emissions would then certainly have been less, as indeed would land use, as disposal of the final ash-containing road waste requires a greater area of land because of the unavoidable mixing with other materials. As the associated costs are probably also higher, this regulatory measure would have negative environmental effects. As yet unspecified, but net positive, effects in comparison with the

alternatives, are the only thing that can shift the balance in favour of use of incinerator ash for road building. (This reasoning should, of course, be substantiated in a more precise, quantified analysis.)

In the context of road building a third alternative is for the incinerator ash to be intricately mixed with all the other road-building materials used, in which case it can no longer be regarded as a separate entity (like fly ash in concrete), with the option of simplifying the double system by making a difference analysis, specifying only those parts of the system that differ between the alternatives. Separate specification of 'only floor ash-related processes' in the road system is then impossible, and waste management of the discarded road in its entirety must be accounted for in the housing-plus-road analysis.

b. Function system comparison

Now the question is how two or more options for delivering a particular function compare, so each alternative is to be specified in terms of this one function only. As an example, we compare a road built without floor ashes and a functionally equivalent road in which incinerator ash is used as a filling material. This involves all sorts of multifunctional processes. We focus our discussion on one such process chain, specifying the boundary between the incinerator and the road system. The standard analysis would be to use the values of the flows involved first for specifying the system boundary for each function delivered to another product system and next for allocation at such boundaries.

One practical problem is that prices are generally lacking when it comes to road waste management. A procedure can then be adopted to 'construct' market prices, as a way of establishing the relative importance of the different outputs in delivering the functional unit 'x kilometres of road'.

The advantages and disadvantages of economic allocation are reviewed in the box below.

Advantages and drawbacks of economic allocation

- The main advantage of this method is that it can be consistently applied to all partitioning situations and for drawing system boundaries in general. However, using an economic parameter for allocation implies basically accepting price structures as they stand, and assuming these to be established in perfect markets. Certain market failures have been addressed above, keeping to the basic price structure as a default approach. The fundamental assumption that neo-classical economics can be satisfactory applied for the purposes of allocation is questionable.
- Particularly in the case of environmental analyses investigating a recycling option or optional use of secondary material, there may be no emerging (recycling) markets with prices on which to base allocation. Generally, the market tends to have a negative perception of these materials, causing a barrier to adequate pricing. In some cases, this perception is initially countered by government subsidies. This may be the case in the example of using demolition waste in road construction (the price would be lower if the market had accepted this alternative and if there was efficient infrastructure for its production). In such cases, production costs can be used to determine the actual price, or the relative material quality can be used as an alternative measure of the economic value of the secondary material.
- Using multi-output allocation for recovered material with a positive economic value suggests that the user takes this co-production into account when deciding to buy the product. This is not generally the case. (Note that even when markets acknowledge the positive economic value of returned glass cullet, say, they do not pay the consumer for it, as the consumer is willing to discard it voluntarily.)