



Contract numbers: 53110/1610, 53110/1620; report completed 31 October 1992

This study was carried out under the auspices of the National Reuse of Waste Research Programme (NOH). The NOH is managed and coordinated by:

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The report consists of two related parts, the Guide and the Backgrounds document, which are only available as a set. Further copies of this report can be ordered from the library of the Centre of Environmental Science (tel. +31 71 277 485). The price is NLG 90.00 per set.

CENTRUM VOOR MILIEUKUNDE  
DER RIJKSUNIVERSITEIT LEIDEN

CIP-GEGEVENS KONINKLIJKE BIBLIOTHEEK, DEN HAAG

Environmental

Environmental life cycle assessment of products / R. Heijungs (final ed.) ... [*et al.*; transl. from the Dutch]. - Leiden : Centrum voor Milieukunde. - Ill.

I: Guide. II: Backgrounds.

Vert. van: Milieugerichte levenscyclusanalyses van producten. - Leiden : Centrum voor Milieukunde, 1992. - Met lit. opg.

ISBN 90-5191-064-9

Trefw.: producten en milieu.

Printed by: MultiCopy, Leiden

Translated by: TechTrans, The Hague

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# PREFACE TO THE ENGLISH EDITION

This is a translation of the original report in Dutch, dated October 1992. Only obvious printing errors have been corrected; new developments have not been included. This preface gives an overview of some developments in the state-of-the-art of LCA since the conception of the original report.

The *Society of Environmental Chemistry and Toxicology* (SETAC) is the current leading international organization in the coordination of the methodological development of life cycle assessment. In April 1993, an expert workshop was held in Sesimbra, Portugal, with the aim of establishing an internationally agreed *Code of Practice*. This included the definition of a technical framework for LCA consisting of components (as in Figure 0.2) and a uniform terminology.

The framework and terminology developed in this report differ slightly from that provisionally developed by SETAC. To avoid confusion we have provided an overview of the main differences here. This is followed by a comparison of the framework and terminology used in this report and that in the *Code of Practice*.

The framework in this report consists of five components. The draft *Code of Practice* consists of four components. The main difference concerns the components *classification* and *evaluation* in the present report. These are part of the *impact assessment* in the SETAC framework. Classification as used in this report is subdivided into *classification* and *characterization* in the *Code of Practice*, where the former denotes the labeling of inputs and outputs according to the effect categories they contribute to, and the latter amounts to the weighting and aggregation into scores for these effect categories. The similarities and the differences between the two approaches are summarized in the table below.

Code of Practice Sesimbra - April 1993	Guide + Backgrounds LCA - October 1992
goal definition and scoping	goal definition
inventory analysis	inventory analysis
impact assessment { <ul style="list-style-type: none"> <li>classification</li> <li>characterization</li> <li>valuation</li> </ul>	} classification evaluation
improvement assessment	improvement analysis

In this study the term *impact* has been avoided. *Interventions* indicate human interference in the environment, e.g. resource extraction and emissions (environmental releases). *Effects* indicate the resulting environmental problems, e.g. resource depletion and acidification. Further differences in terminology are minor.

For the time being this guide is only likely to be used by consultant engineers, large companies and research institutes. A product assessment is relatively complicated to carry out because a lot of process data is not yet available. Once an accessible, good quality database has been created as well as effective software for the method, small and medium-sized businesses and other relevant organisations will also be able to undertake responsible product assessment studies.

# FOREWORD

This is the report of the study *Towards a method for comparative product assessment on environmental effects*. This study was carried out as part of the as part of the Netherlands National Reuse of Waste Research Programme (NOH). The NOH is managed by NOVEM (Netherlands Agency for Energy and the Environment BV) and the RIVM (National Institute of Public Health and Environmental Protection). The NOH is funded by the Netherlands Ministry of Economic Affairs and the Directorate-General for Environmental Management of the Ministry of Housing, Planning and Environment; this department also provides central coordination for the product policy.

The study was mainly carried out by the Centre of Environmental Science (CML) at Leiden University. The research institutes TNO (Netherlands Organisation for Applied Scientific Research) and B&G (Fuels and Raw Materials Bureau) undertook certain parts of the project. We are also grateful for the access we had to the product research expertise of a number of institutes and companies in the Netherlands and abroad. The technical and conceptual aspects of the study were supported by a special think-tank created for this project. A review group set up by the Netherlands Advisory Council for Research on Nature and the Environment (RMNO) monitored coordination with other studies and evaluated the social relevance of the study.

1 The study was carried out between October 1990 and October 1992. During that period interest in environmental product studies grew enormously as did the demand for a guide for product research. For these reasons interim reports were published in March 1991, November 1991 and May 1992. These interim reports provided a provisional guide which reflected the progress of the research. The comments of the think-tank as well as experts in the Netherlands and abroad on these interim reports encouraged discussion on many issues. As a result the final method was developed to a higher scientific standard than it would have been without the interim reports. The provisional, incomplete method was actually applied by several organisations. This not only provided useful suggestions regarding the practicability of the method but also broadened its support in society as well as in industry.

2 The report consists of two volumes, the *guide* and the *backgrounds*. The guide is intended for use in practical studies to assess products on the basis of their environmental effects. The method is made up of *components*; complete elements which provide a result which is useful in itself. The components are built up out of *steps* which provide the procedure of the method. Each step contains a set of *guidelines* to answer common questions. These guidelines are based on theoretical considerations. These considerations are discussed in the backgrounds document. Hence this volume forms the scientific backbone of the guide in which the choices and assumptions made are substantiated by an examination of the theoretical considerations and with literature references.

3 For the time being this guide is only likely to be used by consultant engineers, large companies and research institutes. A product assessment is relatively complicated to carry out because a lot of process data is not yet available. Once an accessible, good quality database has been created as well as effective *software* for the method, small and medium-sized businesses and other relevant organisations will also be able to undertake responsible product assessment studies.

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## INTRODUCTION

This chapter provides a general introduction to environmental product life cycle assessment (LCA). The need for product assessments and the need for a suitable method will be discussed first. The structure of the method presented in this report and how it is linked to the assessment of other aspects will be explained. The chapter ends with a list of the premises defined.

### 0.1 Product assessments

Besides process and substance-oriented environmental policies, the NEPP (Netherlands national environmental policy plan) identifies product-oriented environmental policy as an important area to be developed (VROM, 1989). The process-oriented policy was the first strand in these environmental policies to be developed. This policy is targeted at the actual source. Substance-oriented policy covers all the relevant processes in the chain of production, consumption and waste processing. Product-oriented policy, however, extends to all processes in the *life cycle*, spanning the production, consumption and waste processing chain of a single product<sup>1</sup>. As the substance-oriented and product-oriented policies cover the complete chain of production, consumption and waste processing, the term used to refer to them is *chain management*. Preventive policies use the substance-oriented or product-oriented reference frameworks. The government has indicated various lines of research to develop this product-oriented environmental policy. The DGM-IPC (Directorate-General of Environmental Management - industry, construction, products and consumers department) of the Ministry of Housing, Planning and Environment (VROM) plays an important part in this field.

The issue addressed by product assessments, and therefore by product policy, is that every product and every activity has an adverse effect on the environment. This causes problems: one particular product may produce a lot of waste while another product requires a lot of raw materials. One product results in more emissions in the usage stage while another results in more emissions during the production stage. This hides the overall effect on the environment: all life cycle stages and all environmental effects are important; shifts to other effects or stages in the life cycle must be avoided.

One line of research concerns the development of a standardized method to implement environmental *life cycle assessments* (LCAs) of products. The basic concept behind a life cycle assessment is that the entire life cycle is considered, including all the environmental effects. In this way shifts to other effects or life cycle stages are avoided. To make LCAs relevant to society at large either they should be carried out on the basis of a single standardized method and data on products and processes, or the differences between studies with regard to these aspects should be clearly

<sup>1</sup> The life cycle concept is included in the NEPP but it is not implemented as consistently as in the NEPP-plus (VROM, 1990). Compare measures A 100 and A 100a, for example. This demonstrates that the need to consider entire life cycles is rapidly becoming accepted in environmental policy.

## CHAPTER 0

# INTRODUCTION

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specified.

Life cycle assessments have been developed over the last two decades\*. The initial step was taken in the context of energy analysis in the '70s. Later this was extended by several groups to include a wider analysis of the depletion of resources as well as an analysis based on environmental hygiene considerations. In this way the original delimitation of the subject was removed. Another limitation was that, originally, only a small part of the life cycle of a product was considered, for example, only the stage in which it was used†. Industry and research institutes in particular, were actively involved in these studies, as was the public sector. The supporting theories were largely developed in the context of practical studies.

Both these limitations have been excluded from the life cycle assessments discussed in this report‡ (see figure 0.1). The complete life cycle is considered, i.e. from the extraction of resources to the processing of final waste. An attempt has also been made to draw up and implement a complete list of environmental problems.

materials production		1	
component production		1	
product assembly		1	
product use	2	1, 2	2
processing after disposal		1	
	depletion	pollution	damage

FIGURE 0.1. Environmental product studies can be delineated by environmental effect, e.g. (1), or by life cycle stage, e.g. (2). This study covers the complete range of environmental effects and life cycle stages.

In order to carry out a life cycle assessment three elements need to be developed in parallel:

- *theory*: a standardized method by which life cycle assessments should be carried out;
- *data*: a database accessible to all which contains most, if not all, the process data required for these practical studies‡;
- *practice*: software implementing the theory in a way which can be used by the various target groups and which can also be used to process database information to produce results which are meaningful in terms of life cycle assessments†.

The project discussed here is only concerned with the methods and the theory. There are links with the second stage (the data) in relation to the development of a format for the storage of process data and the selection of relevant data for an environmental assessment. The theory will also affect the way in which a software implementation is made.

This project is not concerned with the development of specific product policy instruments or actual

\* You are referred to Assies (1992) for an historical overview.

† There is, however, no standard division of the stages in a life cycle.

‡ The NEPP-plus states the following with respect to the introduction of an official environmental labelling system (A 101a): "The requirements will relate to all environmental aspects 'from the cradle to the grave'."

§ See also measure A 100c in the NEPP-plus: "In 1991 a start will be made on setting up a database containing information about the environmental consequences of materials and products throughout the whole production cycle." In the feasibility study inspired by this it was found that, despite the needs of the interested parties "a central database [...] cannot be realised from the current position" (Bos *et al.*, 1992). A survey of available databases indicated that there are four files with process data useful for LCAs (Van Oers *et al.*, 1991).

¶ Several such computer programs are currently available. This includes both commercial programs (IIASA, Boustead) as well as experimental demonstration programs (Goedkoop & Jansen, 1991).



policy development. However, it is not always entirely separate from these topics. The reason for this is that choices will have to be made in the overall method, depending on the objective of a given life cycle assessment. For example, depending on the intended application (government policy, business analysis, eco-design, etc.) different choices will have to be made with respect to representative process data, the presentation of the results of the study, the testing on product standards and the way in which products are compared, etc.

For these reasons the study should be linked to possible policy applications and its place in society\*. However, the intention is to develop an "open" method which can be used for a wide range of applications. Hence, later in this report the structure and premises of the method will be defined in general terms, without anticipating specific applications. A short description of the applications (which are not part of the method) in the context of the target groups (consumers, manufacturers, public sector) is included in §1.1.

In recent years many LCAs have been carried out using the general approach described above. Some of these were even for the same product (e.g. milk packaging: BUS, 1984; Franke, 1984; Lundholm & Sundström, 1985; Mekel *et al.*, 1990; Tillman *et al.*, 1991). One observation which stands out when these LCAs are compared is the lack of consistency between them: sometimes the variation between the results is such that different product alternatives are identified as the best in different studies. A further study (Guinée *et al.* 1992a) of the causes of these differences is difficult as many implicit decisions and assumptions were made during the LCAs.

Hence, there are many differences between life cycle assessments carried out over the past few years. Therefore it is difficult to compare the results of these assessments. These differences are partly due to different process specifications; one study might be based on production data for Swedish electricity while another study might use the situation in Germany for the same process. A further complication is that these differences are often due to methodological differences at the general or detailed level and that the premises are not always specified in sufficient detail (Guinée *et al.*, 1992a; Pedersen & Christiansen, 1992).

Generally speaking all these studies trace the entire life cycle of a product. However, there are differences in the way environmental interventions are allocated and in the nature and number of environmental interventions considered. There are also great differences between the ways in which the allocated environmental interventions are aggregated. Therefore the central role which life cycle assessments could play is undermined. As a result these studies will lose their validity if the variation in data and apportionment methods continues to increase. Hence methodological harmonization is a major goal of this study.

This study aims to end differences which are hard to trace. The aims of the study are both to provide a possible solution to the fundamental problems which occur in every LCA and to provide a guideline for specifying the assumptions made when carrying out an LCA. In other words this study provides a *guide* with *guidelines* for the major obstacles to enable LCAs to be carried out in as structured and uniform a manner as possible. The *backgrounds* document provides the scientific basis for the guide. The guide contains compromises between theoretical correctness and practical feasibility. The solutions identified in the backgrounds document may be impossible where data is not yet available or the theoretical aspects have not been developed in sufficient detail. The guide provides a provisional solution for these problems.

The theoretical aspects of environmental life cycle assessments are developing rapidly. This means that some of the concepts developed here may be outdated within months. A step which is correct in principle, but difficult to implement may become feasible, in which event the provisional guideline ceases to be valid. The theory will also become outdated as a result of new information. It is expected that sufficient new material will be available within a year or two to justify a revised edition. At present it is too early to say whether this will actually happen. It is therefore important to specify the date of the method used when implementing an LCA. Furthermore the composition of products and process characteristics will change in the course of time.

\* The NEPP states: "It is important that these projects be carried out in collaboration with private industry and with environmental and consumer organisations."



It will be clear that an LCA does not aim to reveal an objective and eternal truth; its theory as well as the data will need to be reviewed regularly.

## 0.2 Structure

First we will define some terms.

A *life cycle* is the combination of production, use and waste processing processes describing the creation, existence and removal of a tradeable good, i.e. its life "from cradle to grave". A *life cycle assessment* can be used to support a decision about a purchase, innovation of production processes or product approval in the widest sense of the word. Such a decision may be based on environmental, social or economic aspects or other considerations.

An *overall life cycle assessment* is a combination of all the life cycle assessments used to support the decisions referred to above. In German and Swiss literature overall life cycle assessments are referred to as *Produktlinienanalyse*<sup>\*</sup>. A *specific life cycle assessment* is a life cycle assessment concerned with only one of the aspects covered by an overall life cycle assessments. A set of specific life cycle assessments, can be combined to form an overall life cycle assessment if they share a common framework. In practical terms this means that they have a common goal definition, evaluation and application which result in a number of specific life cycle assessments. Figure 0.2 provides a diagram of this.

An *environmental life cycle assessment* is a specific life cycle assessment in which environmental aspects are considered. An *environmental life cycle assessment of products* is an environmental life cycle assessment for the evaluation of products. This is the type of life cycle assessment which will be discussed below. Here, and in other reports, the term "environmental life cycle assessment of products" is often shortened to simply "life cycle assessment" and abbreviated as LCA<sup>†</sup>. Above the phrase "product in the widest sense of the word" was used. This is because products may include services<sup>‡</sup>.

In an LCA as described above the interaction between a given product throughout its entire life cycle and the environment is studied. A clear distinction will be made between the product and the environment. This requires a set of definitions.

The *product system* includes the fulfilment of the product's function and all other processes required for the fulfilment of its function. Main process groups include production, use and waste processing.

The *economic system* consists of all product systems combined. In an LCA the economic system outside the product system in question is also important, it is referred to by the term *other product systems*. The fact that it is important does not contradict the principle of using the life cycle. Because many production processes produce co-products, the use and production of recyclable or recycled waste and the combined processing of waste from a number of product systems means that these other product systems are vital (see also the discussion on multiple processes in §2.1.3).

The *environmental system* consists of all processes occurring in the environment.

Figure 0.3 illustrates the interaction between these three systems. There are five interactions (arrows) relevant to LCAS:

- two between the product system and the environmental system, these are *emissions* and *extractions*;

\* In Öko-Institut (1987) the term overall (*umfassend*) is used in three senses: firstly for the complete life cycle, secondly for all aspects ("Umfassend - im Hinblick auf die zu untersuchenden Konsequenzen in jeder dieser Lebenszyklusphasen auf die Bereiche der Natur, der Gesellschaft und der Wirtschaft.") and thirdly because the function of the product is the central topic. In the first sense life cycle assessments in this project are comprehensive (the vertical axis in Figure 0.1). In the second sense the aim of this project is to cover *all environmental aspects*. The horizontal axis in Figure 0.1 is limited to environmental aspects, other possible specific life cycle assessments in Figure 0.2 are explicitly excluded. The third sense, concerning the function of a product, will be discussed below (see Figure 0.3).

† The abbreviation LCA is not only used for *life cycle assessment* but also for *life cycle analysis*. Other terms include *product life analysis* (PLA) and *quantitative life cycle assessment*.

‡ Strictly speaking we are only concerned with services here, see below.

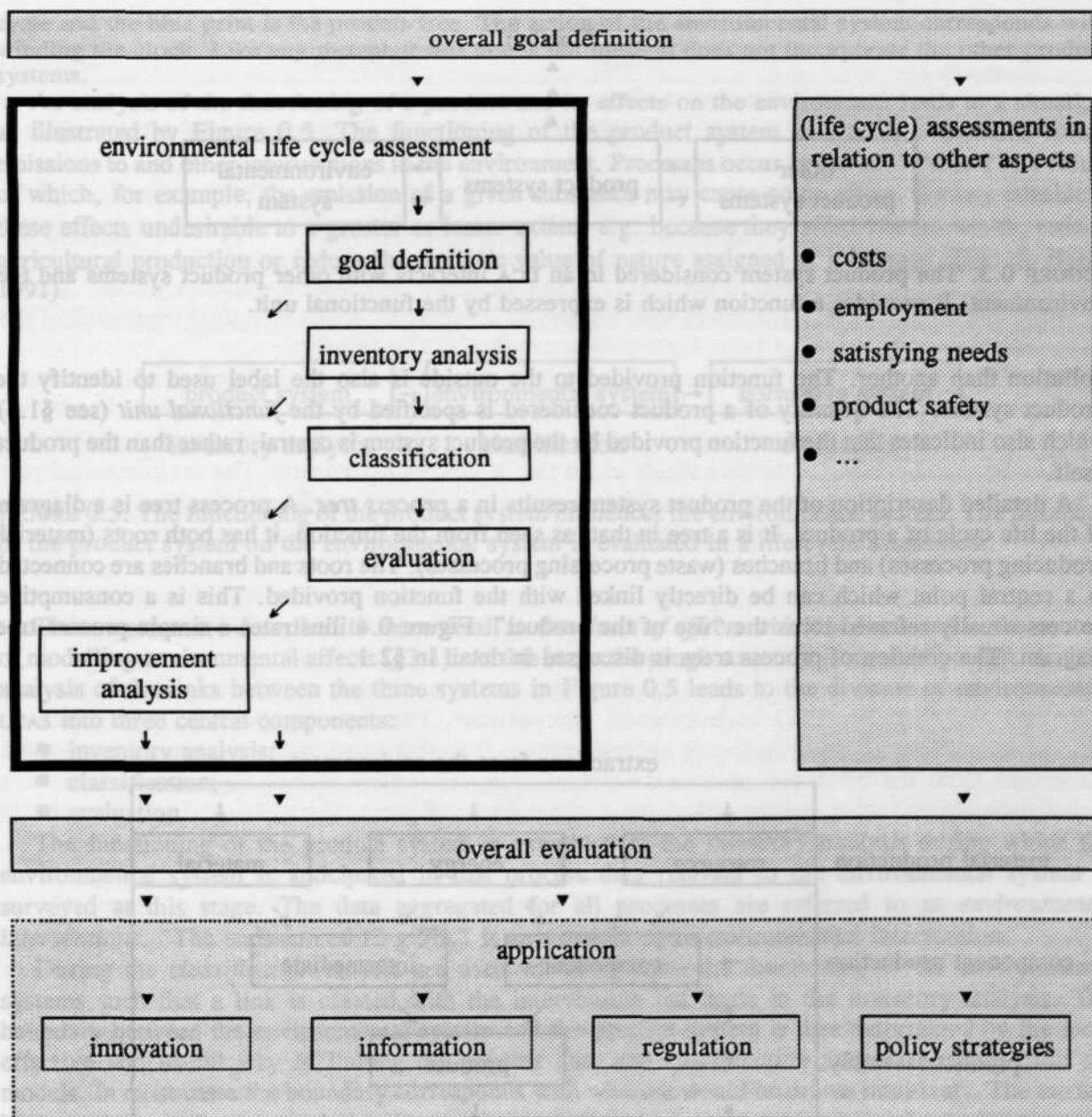


FIGURE 0.2. An LCA comprises the components goal definition, inventory analysis, classification, evaluation and improvement analysis and, with an assessment on other aspects, leads to an application.

- two between the product system and other product systems, these are *use* and *production* of material flows in multiple processes\*;
- the provision of a *function* by the product system to the user.

The function is the *raison d'être* of the product system, it is the only† external result of the product system life cycle and it is therefore the pivot of the LCA. If a result is not specified each product system (viewed as a black box) would only differ from another in that one box would cause less

\* It is open to debate whether the product system interacts with the other product systems or whether this term only refers to allocated process data (see §2.1.3). This discussion is fairly academic in nature, the main conclusion is that the term product system is not always used consistently.

† If co-products and recyclable materials are produced there are clearly other external results as well, but these are supplied to other product systems, not to the user. See also the previous footnote.

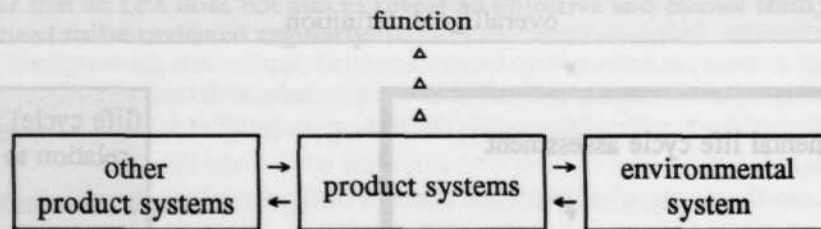


FIGURE 0.3. The product system considered in an LCA interacts with other product systems and the environment. It provides a function which is expressed by the functional unit.

pollution than another. The function provided to the outside is also the label used to identify the product system. The quantity of a product considered is specified by the *functional unit* (see §1.3) which also indicates that the function provided by the product system is central, rather than the product itself.

A detailed description of the product system results in a *process tree*. A process tree is a diagram of the life cycle of a product. It is a tree in that, as seen from the function, it has both roots (material producing processes) and branches (waste processing processes). The roots and branches are connected to a central point which can be directly linked with the function provided. This is a consumptive process usually referred to as the "use of the product". Figure 0.4 illustrates a simple process tree diagram. The creation of process trees is discussed in detail in §2.1.

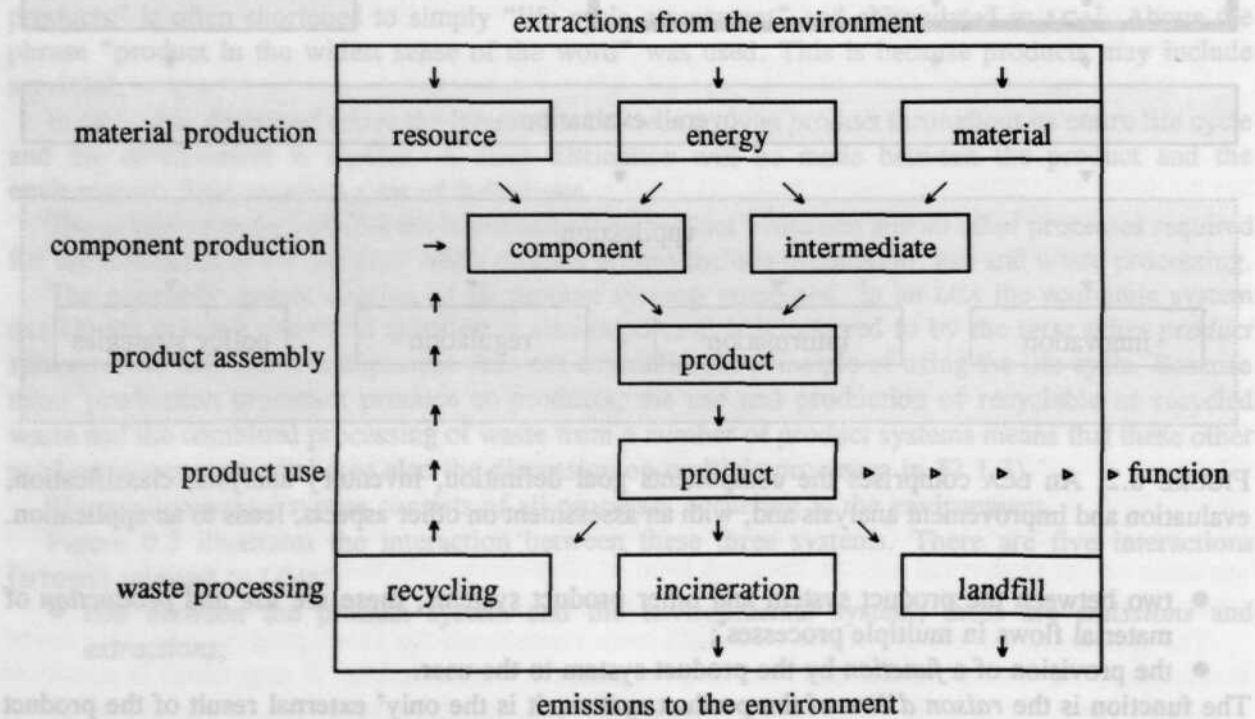


FIGURE 0.4. Simple process tree. For simplicity emissions from production processes, etc. and possible waste flows from each process have not been indicated.

The distinction between product system, life cycle and process tree can be illustrated with a metaphor. A clock converts external energy to a function: winding the clock everyday allows us to check the time. The clock contains a mechanism which runs the clock. The mechanism can be explained with a blue print. In this example the clock is the product system, the mechanism is the life



cycle and the blue print is the process tree. The action of the environmental system corresponds with winding the clock. Like any metaphor this one is not ideal: it does not incorporate the other product systems.

An analysis of the functioning of a product and its effects on the environment leads to a situation as illustrated by Figure 0.5. The functioning of the product system results in extractions from, emissions to and other interventions in the environment. Processes occur in the environment as a result of which, for example, the emission of a given substance may cause some effect. Society considers these effects undesirable to a greater or lesser extent, e.g. because they affect human health, reduce agricultural production or reduce the intrinsic value of nature assigned to it by man (Udo de Haes, 1991).

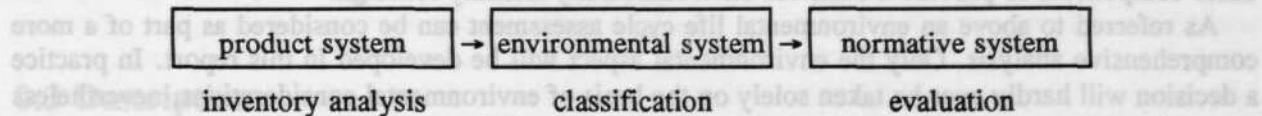


FIGURE 0.5. The functioning of the product system influences the environmental system. The influence of the product system on the environmental system is evaluated in a life cycle assessment.

These matters are described in more detail in the *chain of effects* which is discussed in the context of modelling environmental effects (§3.1). At this stage the rough structure is more important. A clear analysis of the links between the three systems in Figure 0.5 leads to the division of environmental LCAs into three central components:

- inventory analysis;
- classification;
- evaluation.

The functioning of the product system is examined in the *inventory analysis* during which the environmental system is anticipated in that process data relevant to the environmental system is surveyed at this stage. The data aggregated for all processes are referred to as *environmental interventions*. "The emission of 12 g SO<sub>2</sub>" is an example of an environmental intervention.

During the *classification* models are used which describe the functioning of the environmental systems such that a link is created with the intervention list made in the inventory analysis. The boundary between the environmental system and the product system is thus determined by the most effective and useful way of linking the process data and *classification factors* resulting from the models. In most cases the boundary corresponds with where it would be drawn intuitively. The models indicate the contribution made by the product system to some commonly recognized environmental effects. A quantified contribution of this sort is referred to as an *effect score*. "3 m<sup>3</sup> air polluted to the odour threshold" is an example of an effect score.

Finally, in the *evaluation* the environmental effects are appraised in relation to their effects on society, for example. Various effects of the environment on man in terms of his physical and psychological performance, the fulfilment of material needs and nature as an intrinsic unit are compared to establish their priorities. This component was introduced as a clearly separate entity to divorce environmental aspects from social aspects, which are studied in different disciplines\*.

During each stage in the life cycle of a product processes occur which affect the environment in different ways. It was decided to use a set of premises to assess a product on the basis of its environmental effects:

- in the structure of the method a distinction will be made between environmental intervention (inventory analysis) and environmental effect (classification);

\* This distinction is not always made. American literature often distinguishes between *Inventory*, *Impact Analysis* and *Improvement Analysis* (Fava, et al. 1991; Curran, et al. 1991). The *eco-scoring system* (Ahbe et al., 1990; see also Chapter 3) also includes classification and evaluation as a unified component.



- in the structure of the method a distinction will be made between effect on a final variable in the environment (classification) and the target variable on which the assessment is based (evaluation);
- potential effects rather than actual effects will be considered;
- at the same time the aim will be to make an assessment of the environmental effects which is not dependent on a given location.

The last three points will be described in greater detail under the structure of the classification (§3.1).

Besides these three components there is also a framework within which the problem is analysed. This is provided by a component known as the *goal definition*. Goal definition, inventory analysis, classification and evaluation follow each other in this sequence. However not all components need to be completed in every case. The *improvement analysis* is different, it can be started from each of the other components to provide a basis for environmentally-friendly redesign.

As referred to above an environmental life cycle assessment can be considered as part of a more comprehensive analysis. Only the environmental aspect will be developed in this report. In practice a decision will hardly ever be taken solely on the basis of environmental considerations, nevertheless the environmental part will provide a result which can be used independently. The environmental part is organized in five components. Each of these components provides information about a product's effects on the environment, "from cradle to grave". This information is identified by the general term *environmental indicators* (Korenromp *et al.*, 1990; Nieuwenhuis & Korenromp, 1992).

The product to be investigated is defined in the goal definition. Besides the information selected by the researcher (functional unit, which products, etc.) the product's goal definition could also indicate *product properties* such as the potential lifespan and recyclability.

The product system is examined in the inventory analysis. Central to this study is the *inventory table*, the resulting list of all environmental interventions. This list may be very long, over one hundred environmental interventions is not uncommon. It is also possible to derive some aggregated parameters from the inventory table or the process data. Examples of such aggregated parameters include primary and other energy consumption, the quantity of waste, the space used and the quantity of heavy metals introduced into the environment.

The *environmental profile* is drawn up during the classification. It is a quantified list of the contributions made to environmental effects. The main aim of the classification is to provide an impression of the contribution made to environmental problems. In most cases the number of descriptive parameters will be reduced. Normally the environmental profile consists of a handful of effect scores.

The evaluation produces in a comparative judgement or an *environmental index*. This judgement or result is also accompanied by a qualitative, semi-quantitative or quantitative estimate of the validity and reliability of the assessment as a whole.

Finally the improvement analysis provides *starting points* for environmental product improvement.

All the components also provide an opportunity to consider qualitative aspects.

The environmental indicators resulting from a given component in an LCA provide a different descriptive level of the environmental properties of the product system. Combining descriptions at different levels is not informative. An information transfer application is only possible by providing environmental indicators at a single, defined level. Otherwise a bewildering confusion of descriptions may result, and certain aspects may be included two or three times while other aspects may have been left out. For example, in the classification the environmental indicator "energy consumption" is represented as a number of effect scores such as depletion, acidification and the greenhouse effect. Similarly, the extent to which an environmental indicator like "recyclability" can be quantified is questionable (Nieuwenhuis & Korenromp, 1992) and whether this information is relevant in environmental terms is also unclear\*.

Table 0.2 provides a brief overview of the disciplines required for each LCA component and the results of each component.

\* Boustead (1990) writes: "There is a widespread public perception that recycling is a 'good' thing. However it is unclear why." He illustrates this with some recycling scenarios which show the points at which energy efficiency reverses.

TABLE 0.2. Overview of the components, their results and the disciplines required.

component	discipline	results
goal definition	engineering, economics, social sciences	product properties (potential life span, reparability, recyclability)
inventory analysis	systems theory, process engineering	inventory table, aggregated parameters (waste, energy consumption, space use, etc.)
classification	environmental science	environmental profile
evaluation	decision making	environmental index or assessment
improvement analysis	process engineering	starting points for design or redesign

### 0.3 Description

In metrology there are four scales: ratio and interval scales, nominal and ordinal scales (see e.g. Siegel, 1956). The distinction does not lie in the terms associated with each scale but in their algebraic properties:

- *ratio scales* meet the requirements for linear scales (multiplicativity and additivity) and have a fixed, clearly defined zero value (example: kg);
- *interval scales* are based on differences and do not have a fixed zero, thus they are not multiplicative although they are additive. The ratios are also invariant to the choice of another unit or zero (example: °C);
- *ordinal scales* contain a limited number of categories in order of magnitude, hence they are sortable (example: calm, strong breeze, storm)\*;
- *nominal scales* also consist of a limited number of categories but not in any order, this rules out any mathematical operations. Equivalence is the only relationship which can be used (example: mammal, bird, fish).

According to metrology it is possible to move from, for example, an interval scale to an ordinal scale but not the other way round. The first two scales may be considered to be *quantitative* while the last two are *qualitative*. Table 0.3 lists the four scales and their properties.

TABLE 0.3. The four main scales.

type	name	permitted operations
quantitative scale	ratio scale	=/≠, >/<, +/-, ×/÷
	interval scale	=/≠, >/<, +/-
qualitative scale	ordinal scale	=/≠, >/<
	nominal scale	=/≠

Qualitative scales were not avoided in this study but the aim was to change over from quantitative to qualitative scales only at the latest possible stage (preferably only in the evaluation) as the aggregation of quantitative environmental effects is theoretically easier and more meaningful than it is with qualitative environmental effects.

\* The classes of many ordinal scales are defined by numbers. An example is the Beaufort scale in which the numbers 6, 7 and 8 represent a strong breeze, a moderate gale and a fresh gale. Each category also represents a wind velocity in  $\text{m}\cdot\text{s}^{-1}$  which meets the requirements for a ratio scale.

Given the delineation between the components inventory analysis, classification and evaluation the following methods can be distinguished:

- the method in which a qualitative approach is taken in the inventory analysis;
- the method in which the quantified approach is deviated from in the classification;
- the method in which qualitative indicators are used in the evaluation;
- the method in which the evaluation is also quantitative.

Table 0.4 gives the nomenclature of these types of LCA. As the evaluation will eventually result in a qualitative statement which does not represent the scientifically proven effects in the environment but rather a consideration of the social desirability or undesirability of these effects, the distinction between a qualitative and a quantitative evaluation is not as fundamental as the distinction between the other components. Given that the hierarchy of scales does not permit us to go from qualitative to quantitative scales and given the logical sequential order of the three components, there are only four possibilities.

TABLE 0.4. Distinctive properties of the three types of life cycle assessments. This project concentrated on quantitative LCAs (bottom row).

LCA	inventory analysis	classification	evaluation
qualitative	qualitative	qualitative	qualitative
semi-quantitative	quantitative	qualitative	qualitative
quantitative	quantitative	quantitative	qualitative/quantitative

This project was concerned with *quantitative* life cycle assessments, hence we will use a *quantitative* inventory analysis and a *quantitative* classification. The migration to a qualitative statement can eventually be made during the evaluation. Unquantifiable but relevant data will also be considered (see also §2.2.8 and §3.2.19).

#### 0.4 Premises

*The description of nature is not stripped of arbitrariness by naive absolutism,  
but only by recognition and formulation of the points of arbitrariness.  
The only path to objective knowledge leads through conscious awareness  
of the role that subjectivity plays in our methods of research.*

HANS REICHENBACH, *The philosophy of space and time.*

The main premise is that the choices made should be clearly identifiable. A number of premises to be met by an LCA may be formulated on the basis of this principle. The method is further defined on this basis in the chapters of the guide which describe the steps. The premises are:

- choices and assumptions should be clearly specified;
- flexible objective;
- life-cycle-based approach;
- incorporation of qualitative aspects;
- inclusion in a complete product study;
- absolute ontological judgements and relative value judgements;
- minimum set of environmental effects;
- concerned with potential effects;
- *ceteris paribus* and the marginal nature of the activity.



### *Choices and assumptions should be clearly specified*

The aim is to standardize the methodological choices and assumptions wherever possible. Guidelines are also provided to indicate how the user should deal with the choices to be made. The way in which the assumptions made by the user should be identified is also discussed.

### *Flexible objective*

The method is formulated such that specific potential applications are not anticipated. For this reason the method is divided into components. An LCA need not include all components, this will depend on the goal. Each component provides a result which can be used independently; the results are referred to as environmental indicators (see table 0.2).

### *Life-cycle-based approach*

Naturally, a life cycle assessment should be based on a *life cycle*. Hence the complete life cycle of the product, from the extraction of resources right through to waste processing has to be considered. We would like to restate that in this study LCA only means environmental LCA and that only environmental aspects have been considered. By including the term "environmental life cycle assessment" in the title the activity is firmly placed in the policy context of "chain management".

### *Incorporation of qualitative aspects*

In principle the quantitative approach (see §0.3) is used wherever possible, but not to the detriment of unquantifiable qualitative aspects. Guidelines are also provided for dealing with unquantifiable aspects.

### *Inclusion in a complete product study*

An LCA analyses the potential environmental effects of a product. Products could also be analysed on other bases, such as cost, ease of use, effectiveness or safety (e.g. UL listing or TÜV approval). In this project these aspects are only of minor importance and have not been fully incorporated. The effectiveness of a product is sometimes included in the functional unit. In a product assessment in which all the relevant aspects are considered (i.e. an overall life cycle assessment) these aspects should also be considered individually.

### *Absolute ontological judgements and relative value judgements*

An LCA does not produce absolute qualitative ratings such as "good" or "environmentally-friendly" but instead qualitative or quantitative statements. Examples include the interventions in the inventory table (" $x$  kg CO<sub>2</sub> emission"), effect scores in the environmental profile (" $x$  units of potential acidification"; both are examples of absolute ontological judgements) or the statements made in the evaluation ("better than product X"; an example of a relative value judgement).

### *Minimum set of environmental effects*

The types of relevant environmental interventions covered by the study are not governed by any rules. However, there are minimum requirements for the main categories of environmental effects: depletion due to the extraction of resources, pollution due to emissions of potentially harmful substances and damage due to space use by, for example, final waste.

### *Concerned with potential effects*

The classification is not concerned with the actual effects of the environmental interventions on ecosystems, rather the environmental effects are defined at a higher level of abstraction. In this way area-specific information which is irrelevant to the product policy can be made abstract. The actual environmental effects can be approximated more closely by using a more area-dependent assessment. This lies more within the domain of environmental impact assessments (EIA) or risk assessments for the choice of the location of a given plant.



*Ceteris paribus and the marginal nature of the activity*

The *ceteris paribus* principle is used in product assessments: the choice of the functional unit of the product alternative investigated should not influence other activities on this planet. For example this means that the effects of emissions due to the functional unit are assumed to be additional to the normal background concentration. The functional unit is also assumed to be marginal relative to other activities so that linear models can be used.

The method is based on the assumption that the environmental effects of the product are additional to the background concentration. This means that the effects of emissions due to the functional unit are assumed to be additional to the normal background concentration. The functional unit is also assumed to be marginal relative to other activities so that linear models can be used.

Table with 2 columns: 'Aspect' and 'Incorporation of qualitative aspects'. The table lists various aspects of environmental impact and how they are incorporated into the LCA process.

An LCA analysis the potential environmental effects of a product. Products could also be analysed on other bases, such as cost, ease of use, effectiveness or safety (e.g. UL listing or TÜV approval). In this project, these aspects have not been fully investigated. The effectiveness of a product is measured in the functional unit. In a product assessment, it is assumed that all these aspects are considered. In an LCA, these aspects should also be considered individually.

Absolute ontological judgments and relative value judgments. An LCA does not produce absolute qualitative ratings such as "good" or "environmentally friendly" but instead qualitative or quantitative statements. Examples include the intervention in the inventory table ("CO<sub>2</sub> emissions") effect scores in the environmental table ("a unit of potential acidification") and an example of absolute ontological judgments (or the statements made in the evaluation (Table 1) as an example of a relative value judgment).

The types of relevant environmental interventions covered by the study are not governed by any rules. However, the main categories of environmental effects are: damage due to space use, for example, final waste, which should be taken into account; damage due to space use, for example, final waste, which should be taken into account.

- Concurrent with potential effects
- The classification is not concerned with the actual effects of the environmental interventions on ecosystems, rather the environmental effects are defined as a higher level of abstraction. In this way, use-specific information which is irrelevant to the product policy can be removed. The actual environmental effects can be approximated by using a more general assessment.
- This lies more within the domain of environmental impact assessment (EIA) or risk assessment for the choice of the location of a given plant.
- effects related with resources
- effects related with the nature of the activity.

## CHAPTER 1

# GOAL DEFINITION

The purpose of the life cycle assessment is determined in the goal definition. This includes not only the subject of the study but also aspects such as the target group, desired application, etc. The goal definition will largely be the result of discussions between the client, researchers and those supporting the study and will also depend on factors such as the time and money available as well as the available process data. The options and limitations on life cycle assessments and the items to be considered when defining the goal can only be discussed here in general terms.

The guide distinguished three steps in the goal definition:

- determining the application (step 1.1);
- determining the depth of the study (step 1.2);
- defining the subject of the study (step 1.3).

Most of these topics are developed in the guide. This chapter contains a section about the use of the life cycle assessment as a product policy instrument, as well as a section on choosing the functional unit.

### 1.1 LCA applications

A life cycle assessment is an instrument to support decision making. There are four main groups of applications:

- product information
- product regulation
- policy strategies
- product innovation.

#### 1.1.1 Product information

LCAs have so far been used mostly to collect and disseminate information about the environmental effects of the functioning of products, the comparison of product alternatives being the most common form. Numerous studies, ranging from the popular to extremely thorough, have been carried out to compare a number of different products. Consumers and consumer organizations as well as the environmental movement have a clear need for information about environmental aspects given the dilemmas they face when purchasing products. While the public and private sectors want to meet this need by providing that information.

We should be wary of using a life cycle assessment as an absolute measure. Given the problems involved in defining the boundary of the system, with the allocation rules, due to the lack of a complete list of environmental problems and so on, it is impossible to make an absolute statement about the environmental acceptability of a product. It is also doubtful whether this will be possible in the future. One of the activities in a life cycle assessment is a sensitivity analysis (§4.2). One of the

aims of this is to determine whether the choices made with respect to the above problems will affect the conclusion. This means that the exact location of the system boundaries should not influence the award of an eco-label, for example. An absolute LCA will not result in a conclusion in terms of "poor" or "better" but in numerical information. All that a sensitivity analysis can add to this is further information about these figures, such as an estimated error margin. However, this will be less relevant than a sensitivity analysis in a product comparison.

### 1.1.2 Product regulation

Regulation of the use of certain products is one aspect of product policy. One of the options for using an LCA is to assess products on the basis of a standard. Assessing products is also a form of comparison by determining whether a product does better or worse than the standard<sup>\*</sup>. The major main is that a normal comparison covers two or more product alternatives. There are at least two types of standards: *official environmental approvals* and *product standards*. In this context an environmental approval refers to a requirement which a product has to fulfil to be eligible for a "green stamp". A product standard, however, can also be used to issue "red stamps": a product may not be allowed onto the market because it fails to meet certain requirements. Both types of standards will have to be defined for specific product groups rather than in universal terms. This is because the intention is to make a distinction *within* a product group, rather than a distinction *between* product groups. The question automatically arises whether certain product groups should be excluded from the environmental approval system as they are too environmentally unfriendly, or, in even more extreme cases, whether complete product groups are unable to meet the product standard. Another problem with a standard is the aggregation level used in the evaluation. A standard can be defined at the level of the environmental index, i.e. after a weighted evaluation. In this case the comparison will be easy: if the environmental index of a specified functional unit of a product exceeds the standard the product does not meet the requirements. However, the standard could also be defined as an environmental profile or even as an inventory table. In this case acceptance of the product or the award of an eco-label would depend on the way in which the standards are specified: should the product meet the standard in all respects, in most aspects or only in relation to one aspect? Thus life cycle assessments can be used for environmental approvals or to set product standards but the above reservations should also be taken into account.

### 1.1.3 Policy strategies

Scenario studies are a third application of comparative assessments. Instead of comparing two products, two or more situations analysed in which the market shares of these products differ. A scenario study for milk packaging is a relatively straightforward example. In this case a certain level of milk packaging consumption is taken as a basis and the distribution among different packaging alternatives is varied. A project on a larger scale could include a scenario study for a complete sector or even a macroeconomic activity. In principle different transport policy scenarios could be analysed with an LCA. It is to be expected that such studies will be undertaken once a good database is available<sup>†</sup>.

### 1.1.4 Product innovation

Product innovation is the last application discussed here. Figure 1.1 illustrates a possible product innovation procedure.

This figure shows that a life cycle assessment of the product to be improved is made first. This could focus on an important environmental aspect. The method described in Chapter 5 can be used to identify the options for improvement. Those carrying out an LCA could make an initial selection

\* This standard is generally defined in abstract terms. However, a "standard milk packaging unit", with which all products are compared, could be considered.

† The report *Our common future* (WCED, 1988) includes the following recommendation: "A broader environmental assessment should be applied not only to products and projects, but also to policies and programmes, especially major macroeconomic, finance and sectoral policies that induce significant impacts on the environment."



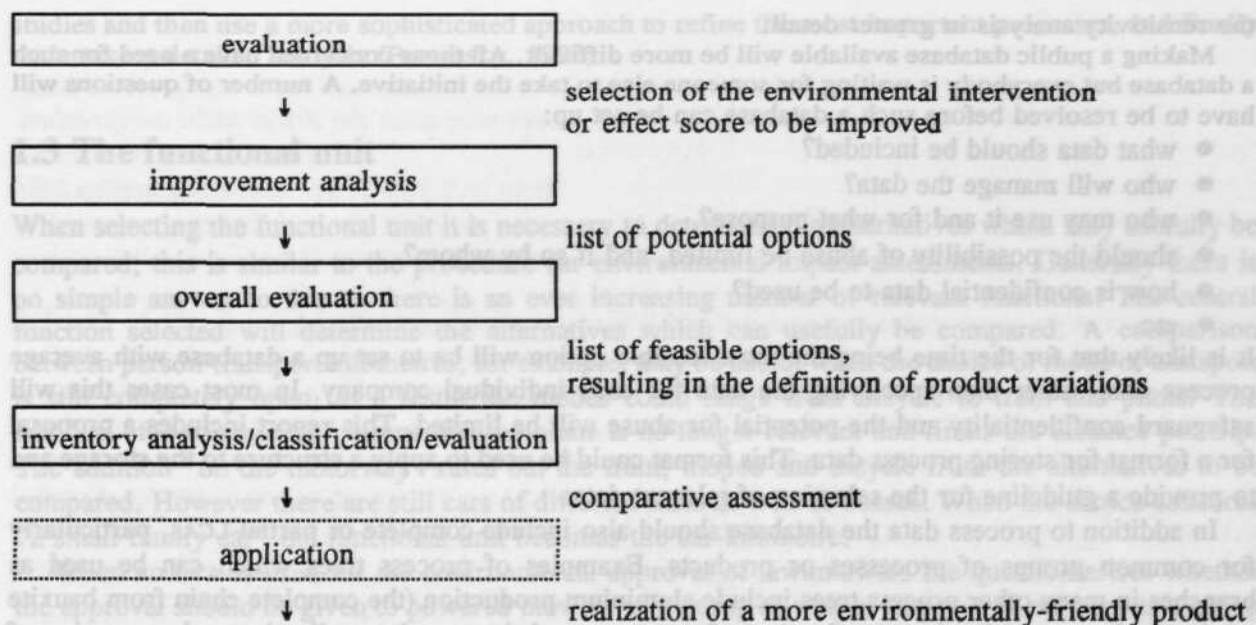


FIGURE 1.1. Product innovation procedure: example of the component sequence in a life cycle assessment which will eventually produce a more environmentally-friendly product.

from these options based on their practical experience. These options are then assessed on the basis of their technical and financial feasibility and other aspects in an overall evaluation. Finally, this will result in the definition of several product variations. These are then compared with each other and with the original product in a comparative LCA. This may lead to a further analysis and the process may be repeated several times. Ultimately one of the product variations will be adopted.

## 1.2 Streamlined LCA methods

The method described in this report is based on a *complete LCA*<sup>\*</sup>. As illustrated in Figure 0.1 a complete LCA covers all stages in the life cycle and all environmental effects to prevent environmental problems being shifted to other stages or effects. A considerable investment in time is required to undertake a complete LCA: three person-months is quite normal for an LCA for which considerable process data are available and one person-year is not unusual for complex studies where the data is difficult to obtain. This is an obstacle to all LCA applications: for the implementation of product policy, when using an LCA in the design process, etc. To make LCAs more widely applicable there is an urgent need to find a more streamlined approach. There are two ways of approaching streamlining:

- software as well as a large public database with process data will have to become available;
- more rapid, less detailed procedures will have to be created.

Both solutions are now being put into practice.

Various computer programs have recently been written to support the implementation of LCAs. At CML the old version of the method was used to develop the SIMAPRO 1.0 program (Goedkoop & Jansen, 1991). IIASA described and developed the IDEA program (Lübker *et al.*, 1991) and Boustead supplies an extensive database with a program. Other programs, which are not widely available have also been developed. It is to be expected that the method described in this report will eventually be implemented in software. Some of the results of the study could easily be implemented as they have been developed in the form of formulas. The reliability analysis and the marginal analysis in particular are so complex that the development of a new computer program would be an obvious way of developing

\* The term "completeness" does not include non-environmental aspects; these aspects are covered by the term *overall LCA*.

the sensitivity analysis in greater detail.

Making a public database available will be more difficult. All those concerned have a need for such a database but everybody is waiting for someone else to take the initiative. A number of questions will have to be resolved before such a database can be set up:

- what data should be included?
- who will manage the data?
- who may use it and for what purpose?
- should the possibility of abuse be limited, and if so by whom?
- how is confidential data to be used?
- etc.

It is likely that for the time being the most feasible option will be to set up a database with average process data rather than specifying the data for each individual company. In most cases this will safeguard confidentiality and the potential for abuse will be limited. This report includes a proposal for a format for storing process data. This format could be used to apply a structure to the storage and to provide a guideline for the selection of relevant data.

In addition to process data the database should also include complete or partial LCAs, particularly for common groups of processes or products. Examples of process trees which can be used as branches in many other process trees include aluminium production (the complete chain from bauxite mining to the material), the use of means of transport, and the complete collection and processing of household refuse. However, this will only be useful once the development of the inventory analysis method has been completed.

Even when all data of the required quality is available an LCA will still take a long time. The reason for this is that the system boundaries, the structure of the process tree, the processes to be selected from the huge data file, etc. will still have to be determined in each case. There will still be a need for a more rapid method even when the primary pre-condition has been met. In the streamlined method the method described in this report is not followed completely and some aspects of it are different.

In principle there are two ways of developing a streamlined method:

- simplify the complete theory;
- develop rules of thumb based on practical experience.

The first approach is also known as the *top-down* approach while the second is known as the *bottom-up* approach. There are at least two sources of expertise for the second approach: those carrying out LCAs as well as designers and process engineers. The first group, for example, knows from experience which processes contribute little to the environmental profile. The second group is mainly familiar with aspects such as low energy consumption, good recyclability, etc.

The following limitations could be considered when developing streamlined methods:

- analysing only the differences ;
- including fewer processes;
- including fewer environmental effects and thus making an inventory analysis of fewer environmental interventions;
- omitting one or more components or steps;
- etc.

The development of streamlined methods falls beyond the scope of this project. However, some simplifications already used in practice have been employed. We did not aim for completeness when defining the system boundaries, on the contrary we considered the need for simplification. As a result several guidelines for this have been drawn up. Furthermore environmental effects for which data is not available in practice, e.g. radiation, have not been included. In a few years' time it should be possible to develop rules of thumb and simplifications based on practical studies. Preferably these should be appraised using a case-specific sensitivity analysis. It will also be worthwhile to try to find existing and new simplifications, to investigate their relevance to specific applications and their reliability and then to combine them in a simplified guide for streamlined LCAs.

As far as the current practical implementation is concerned we recommend that the results and considerations from other studies be used for inspiration and to develop one's own view. An iterative, stratified approach could also be used: first define the process data based on estimates and earlier

studies and then use a more sophisticated approach to refine the most important processes, and finally undertake a detailed analysis (Poremsky, 1992).

### 1.3 The functional unit

When selecting the functional unit it is necessary to determine the alternatives which may usefully be compared; this is similar to the procedure for environmental impact assessments. Generally there is no simple answer to this as there is an ever increasing number of relevant functions. The central function selected will determine the alternatives which can usefully be compared. A comparison between person-transport kilometres, for example, may be useful when the choice of mode of transport is still completely open, as a result the modes could range from bicycle to train and plane. The addition "for commuting" means that the plane is no longer relevant and limits the distance per trip. The addition "on the motorway" rules out the train, moped and bicycle from the alternatives to be compared. However there are still cars of different sizes as well as busses. When the choice concerns "a small family car" the functional unit becomes the car-kilometre.

When an LCA is used for the environmental approval of lawnmowers the question arises whether the approval should be given to powered mowers as a group as well as unpowered mowers as a group or whether mowers should only be selected on the basis of their function "ability to mow".

Fresh milk provides another example. When the functional unit "the consumption of 1000 litres fresh milk" is selected the production of the milk, its transport, processing, packaging, distribution and storage at home before consumption have to be included in the calculation as well as the sewage treatment, transport back to the factory and processing after disposal. Many of these processes are irrelevant to the choice of packaging system as they are identical for all the packaging systems under consideration. The volume of milk to be produced does not depend on the choice of packaging. However the fact that the volume of milk to be consumed will be reduced needs to be taken into account because more milk may be left in a carton than in a bottle. This may be different for custard. In this case the amount of custard will have to be considered. There are also functions which have more to do with a gesture than the quantity. Flowers provide a good example of this: people generally want "a bunch of flowers", rather than "flowers for ten days" or "500 gr of flowers".

Hence, the functional unit should always be defined with the end-user in mind. Leaving out elements which are identical in all the alternatives considered will affect the way in which a formalized overall assessment can be undertaken (see Chapter 4). This is also another reason why it is impossible to draw up an absolute ratio (or any other scale) of environmental-friendliness.

The functional unit should be defined as accurately as possible, with the proviso that the selected products should be covered by the definition and that the functional unit is compatible with the nature of the application. The definition of a suitable functional unit is difficult, particularly when making recommendations for future innovation. If the functional unit included the phrase "fitted with a bright yellow cap" it would be difficult to implement an improvement measure to reduce cadmium in that functional unit.

Wherever possible the functional unit should be defined as a *service* to the user or to a process. Thus the functional unit should describe a use-function whenever possible. This means that the selected products must have an independent function either for a consumer or for a process. As a result it is difficult to think in terms of the life cycle of a material. The material cannot be described "from the cradle to the grave" as this would require the development of all applications. However, when the function of a material in a product is defined, a *partial LCA* covering the role of that material in the product will be possible. Nevertheless, the application in actual cases will determine how the "complete" comparison and assessment of two alternative materials works.



## 1.4 Discussion

*Zu einer Antwort, die man nicht aussprechen kann,  
kann man auch die Frage nicht aussprechen.*

*Das Rätsel gibt es nicht.*

*Wenn sich eine Frage überhaupt stellen läßt,  
so kann sie auch beantwortet werden.*

LUDWIG WITTGENSTEIN, *Tractatus logico-philosophicus*.

Unlike most other components the goal definition is not predominantly technical but mainly social in nature. This is also confirmed by the three steps given in the guide: these mostly involve discussions between parties and how these are laid down. It is not possible to provide a definite guideline as to what constitutes a "good" functional unit. When two parties disagree about the choice of a functional unit it is likely that both their arguments have points in their favour. The same applies when selecting the product alternatives to be considered. In a comparative LCA of writing paper it may be disputed that recycled paper is always a suitable alternative for all paper uses. For this reason keeping an open mind is particularly important during the goal definition. It should be clear which alternatives have been considered and which were excluded and the reasons why. Only then will someone using the results of the LCA be able to decide whether they agree with the considerations and whether recycled paper would be a suitable alternative for their particular application.

Not only is the choice of alternative debatable, but the inclusion of the *zero alternative*, i.e. a situation in which there is no consumption (Öko-Institut, 1982), could also be advocated. This is not a particularly interesting option in a product comparison but the opportunity to discuss the function consumed should still be seen as positive. Almost any product, irrespective of its environmental profile, can be identified by an LCA as the best if the functional unit has been strictly defined. Clearly this should never be the objective of an LCA: an LCA should be used for analysis and assessment, never as a justification. Arguments such as "it is still better than flying" or "this product only contributes 1% to the depletion of the ozone layer" are not acceptable.

In view of the potential influence on consumer purchasing it is important that an independent body be set up to supervise the careful implementation of LCAs for general use which could also mediate in disputes. Steps towards the creation of a code of conduct, complete with standardization of the terminology and methodological aspects, are now being taken at international level.

## CHAPTER 2

# INVENTORY ANALYSIS

The inventory analysis may be defined as the component of an environmental life cycle assessment in which the product system is analysed. The external result of the product system (the function in Figure 0.3) was one of the items defined in the goal definition. The internal part of the product system is not relevant to this function. However, the external aspects cannot be fulfilled unless the internal aspects function properly. The internal "mechanism", i.e. the life cycle, consists of a range of processes and has various side effects: the environmental interventions. These environmental interventions are determined in the inventory analysis by revealing the internal structure. The environmental interventions of the external unit equals the sum of the environmental interventions of the internal units.

Thus the internal structure of the product system is determined by processes. These are exclusively economic processes; all processes occurring in the environment and their effects on the environment are analysed in the classification. An economic process can be considered as an economic activity which transforms a given input from the economy and the environment into an output to the economy and the environment. Products, services or resources, etc. are converted to other products, services or emissions. All products and services used are supplied by another process and all products and services (particularly waste to be processed) produced are used by other processes. These supplying and consumption processes are by definition included as part of the product system when the complete life cycle is considered. The only exception to this is the process which provides the required function as a service.

The processes which constitute the life cycle can be arranged in the form of a process tree. A process tree is a systematic arrangement of the processes which make up the product system. The characteristics of each process included in the process tree have to be determined. Ready-made process specifications are available for some processes, such specifications may have been compiled and reported in earlier research based on the manufacturer's data, on an application under the environmental regulations or on an emission register. A lot of data will have to be collected through literature studies, surveys or on a knowledge of process engineering. The reliability of the data collected can be checked by comparison with several sources. The collected process data should relate to actual or possible processes wherever possible. The least possible number of assumptions and simplifications should be made.

The simplifications require special consideration: a comparison of different sources as well as the use of verification aids such as mass and energy balances is made much more difficult when simplifications were used when collecting the process data. This would also hinder new developments and insights. For example, CO<sub>2</sub> emissions were generally not included in process data collected in the '70s. This was to avoid making the tables unnecessarily long, at that time CO<sub>2</sub> was not considered to be important. The use of old data together with later ideas about the greenhouse effect necessitates the reconstruction of CO<sub>2</sub> emissions using mass balances or the estimated carbon content of fossil fuels, for example. Another common simplification is the reduction of multiple processes to single processes.

The production of chlorine and sodium hydroxide by electrolysis of common salt is an example of a multiple process. Many process data files include a process which produces chlorine without also producing sodium hydroxide. Similarly there may be another process which only produces sodium hydroxide. Thus, in many studies the actual multiple process has been divided into a number of fictitious single processes. This may make the process data easier to comprehend but it is often unclear how the division was made. Furthermore this may have affected the mass and energy balances which means this verification aid can no longer be used.

This also provides an introduction to the next step in the inventory analysis: the division (allocation) of multiple processes. The following aspects of each process have to be specified: whether it is a single or multiple process, its useful products and the way in which allocation should be made. It is then possible to replace the multiple processes in the process tree by single processes.

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 what is required and all that remains is a service which is provided externally in the form of a functional unit. The factors by which the processes were multiplied are also applied to the environmental interventions. Their aggregation across all the processes produces the inventory table.

Further to the above the following four steps are identified in the guide:

- drawing up the process tree (step 2.1);
- entering the process data (step 2.2);
- application of the allocation rules (step 2.3);
- creating the inventory table (step 2.4).

The specific problems associated with these steps, which are developed in the guidelines, will be discussed here. The formulas used to determine the occurrence of each process will also be derived.

## 2.1 The system boundaries

Figure 0.3 already illustrated the interaction between the product system under consideration and other systems. The other systems referred to are the environmental system and the set of all other product systems. The following interactions occur at the intervention level (see Figure 2.1):

- process which are only partly included in the product system (*a*, *b*, *c* and *d*);
- processes which are not included in the product system (*e* and *f*);
- extraction from and emission to the environmental system (*g*).

The processes which are considered as only partly included in the product systems can be divided in two types:

- multiple economic processes (*a* and *b*);
- reuse and recycling (*c* and *d*).

The exact delineation between the product system and the other two systems is discussed in the following four sections. The delineation between the economy and the environment (problem *g*) is described in §2.1.1, the omission of processes is discussed in §2.1.2 (problems *e* and *f*), §2.1.3 covers the allocation of processes or chains of processes which serve two product systems (*a* and *b*) and the problems associated with reuse and recycling (*c* and *d*) are discussed in §2.1.4.

### 2.1.1 The environment and the economy

When creating the process tree it has to be decided which processes should be included in the product system and which in the environmental system (*g* in Figure 2.1). This might appear to be trivial but a number of problems may arise. Agriculture can be included in the product system and similarly a production forest is part of an economic system. It becomes more difficult when a natural forest and a production forest merge. A decision will have to be made in such cases. However, in practice the choices made will have little effect on the end results. This can be tested with a sensitivity analysis (part of step 4.2).

The situation where certain processes are included in the environmental system for practical reasons



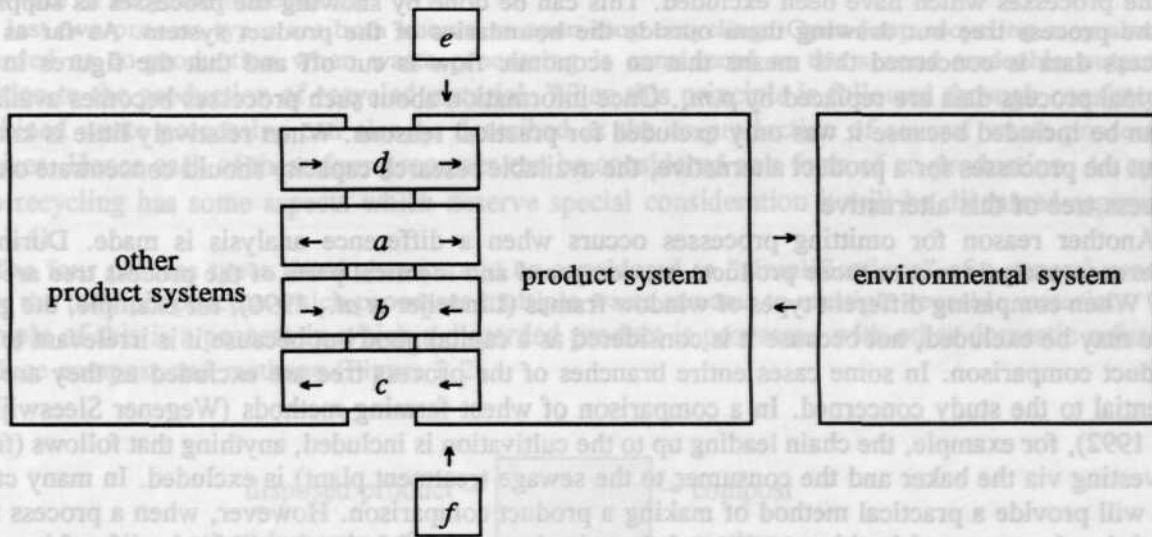


FIGURE 2.1. The processes to be included in the product system are determined during the inventory analysis. Decisions have to be made about the environment/economy, cut-off points and about multiple production processes.

is of a different nature. While information about the long-term leaching of toxic substances from landfill sites to the soil and groundwater is not clear, solid waste could be considered as an emission combined with space use (see also §2.2.7). In most of the studies carried out so far solid waste has indeed been considered as an environmental intervention. In that event solid waste is included as an independent parameter in the classification and may be classified on the basis of its hazard (chemical, radioactive, etc.). When emission data are available landfill can be considered as part of the product system. It will still have a space use but the hazard categories can now be interpreted as emissions, or the risk of emissions, of certain pollutants.

### 2.1.2 Completeness

It will always be necessary to create a boundary between processes which are included in the process tree and processes which may reasonably be excluded. This concerns the processes *e* and *f* in Figure 2.1. For example, these processes may represent the use of capital goods, both for production (e.g. depreciation of machines) and for disposal (e.g. depreciation of sewage treatment plants). This will often include the infrastructure of processes which provide a service (e.g. road construction). These processes are not allocated to other product system, instead they are included in the process system under consideration. However, they are not quantified in the process system but included as *p.m.*\*

It is difficult to provide exact criteria for the elements which may reasonably be excluded†. The use of a simple numerical criterion, e.g. excluding all processes whose contribution to any environmental aspect is less than 1%, may have undesirable consequences. This could result in almost all environmental effects being excluded when the processes are divided into their constituent sub-processes. Furthermore, comparisons would also be distorted if this was only done for one of the

\* Abbreviation of the Latin phrase *pro memoria* (as a reminder).

† The addition of new processes which are also relevant, will raise the total intervention score to a limit. The reason for this is that there is only a limited number of environmental interventions and products in the world to which they have to be allocated. In principle it should therefore be possible to calculate these limits based on a series of additions. However, the lack of "standard units of addition" means that for the time being such a procedure is not possible.

product alternatives\*. To make a responsible choice in this matter requires at least a general overview of the processes which have been excluded. This can be done by showing the processes as suppliers to the process tree but drawing them outside the boundaries of the product system. As far as the process data is concerned this means that an economic flow is cut off and that the figures in the original process data are replaced by *p.m.*. Once information about such processes becomes available it can be included because it was only excluded for practical reasons. When relatively little is known about the processes for a product alternative, the available research capacity should concentrate on the process tree of this alternative.

Another reason for omitting processes occurs when a difference analysis is made. During a difference analysis two or more products are compared and identical parts of the process tree are left out. When comparing different types of window frames (Lindeijer *et al.*, 1990), for example, the glass pane may be excluded, not because it is considered as a capital good but because it is irrelevant to the product comparison. In some cases entire branches of the process tree are excluded as they are not essential to the study concerned. In a comparison of wheat farming methods (Wegener Sleeswijk *et al.*, 1992), for example, the chain leading up to the cultivation is included, anything that follows (from harvesting via the baker and the consumer to the sewage treatment plant) is excluded. In many cases this will provide a practical method of making a product comparison. However, when a process tree which has been pruned in this way is used there is always the risk that it will lead a life of its own, and that it will not be clear that a section has been omitted. Once the inventory table has been calculated it can be used as a supply service in another process. When "incomplete functional units" are used environmental interventions will be underestimated. Confusion about complete and incomplete inventory tables can be avoided by using the format for the product system as well (see §2.3). The distinction between complete and incomplete process trees is then clear from the economic inputs and outputs which will be equal to, or not equal to, zero.

### 2.1.3 Allocation rules

Process quantification requires taking a number of methodological decisions. An important methodological choice is made or applied in step 2.3. This concerns the processes associated with a functional unit which are not only included in the product system under consideration but are also part of another production system (*a*, *b*, *c* and *d* in Figure 2.1).

The crux of the matter is that only a limited part of the process is applied to the functioning of the product system under consideration and that the rest of the process serves another product system. These are processes which produce several useful outputs, which are known as *multiple processes*. When only one of these useful outputs is required for a given product system the multiple output process is essentially split into a number of *single processes*. The environmental interventions of the multiple process and the economic flows which are not the aim of the process are distributed among the single processes. This distribution is known as *allocation*†. There are many notions and misapprehensions about allocation, however there is no intrinsically correct method of allocation. In this section we will attempt to structure the discussion on this subject.

Figure 2.1 illustrates four types of processes where the allocation issue may be relevant: these are all multiple processes. As seen from the product system they can be described as:

- co-production of multiple products (*a*);
- combined waste processing (*b*);

\* In some methods this is done specifically for rapid product assessments, one of the reasons is to save time (see e.g. Korenromp *et al.*, 1991). For example, in a comparison of incandescent lamps and energy-efficient lighting the energy consumption during production might be included for the energy-efficient lighting but not for incandescent lamps as the energy consumption of incandescent lamps while they are used is much greater.

† Some authors suggest that it is possible to dispense with such a distribution. In many cases the general system requirement, that the sum of the fictitious processes should equal the original process, is not fulfilled. The description in terms of a number of equivalent processes is an example of this (see ILV, 1991). Similarly the introduction of multiple functional units (Heintz & Baisné, 1992) does not appear useful as the functional unit is then expanded with other functions which can be fulfilled due to the presence of a byproduct, as a result of which the assessment may expand enormously; in the end almost all processes on earth are linked by one process or another.

- production of recyclable material (c);
- use of recycled material (d).

The last two process types are both known as open-loop recycling. Open-loop recycling may also be regarded as co-production when waste processing is considered as the second tradeable output in addition to the production of recycled material. When this principle is followed through consistently combined waste processing can also be described as the co-production of several waste processing services. Hence each of these four processes can be considered as a form of co-production. As open-loop recycling has some aspects which deserve special consideration it will be discussed separately (§2.1.4).

The four process types listed above could be considered as "simplifications" of a general process type: the multiple process which processes multiple waste streams to multiple reusable materials. One example of this is a process in which a discarded product is processed with other domestic refuse to produce compost and methane (Figure 2.2).

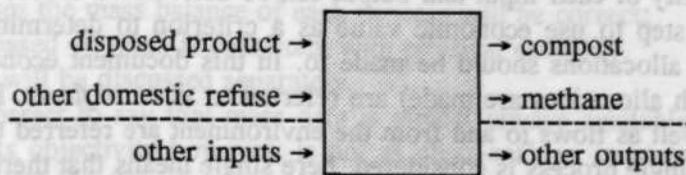


FIGURE 2.2. Most common form of the allocation issue: a process processes multiple waste flows to multiple useful materials. The associated interventions are allocated to the four flows by allocation.

Two questions arise when allocating multiple processes (ILV, 1991):

- the qualitative question: what is allocation to what?
- the quantitative question: how is the allocation made?

These questions can be expanded into:

- which flows are allocated?
- which flows are allocated to?
- which flows are allocated to which flows?
- to what extent is each flow to be allocated to the various other flows?

We will use the process which produces the economic outputs chlorine and caustic soda and which emits  $\text{NO}_x$  and  $\text{HCl}$  as an example. The four questions could be answered as follows:

- $\text{NO}_x$  and  $\text{HCl}$  are allocated;
- they are allocated to chlorine and caustic soda;
- $\text{NO}_x$  is allocated to both chlorine and caustic soda,  $\text{HCl}$  is only allocated to chlorine;
- $\text{NO}_x$  is allocated to chlorine and caustic soda in a ratio of  $1 \div 2$ .

The first question to be answered is: which flows are to be allocated and what should they be allocated to?

#### *What is allocated to what? Main flows and subflows*

Given the general process structure there are four typical flows in a process:

- an economic input (goods, services, materials, energy and waste to be processed);
- an input from the environment (resources and space);
- an economic output (similar to the input);
- an output to the environment (emissions, etc.).

Inputs from and outputs to the environment are free or have a negative value: fees sometimes have to be paid for the extraction of resources and levies are sometimes due for discharges; however the extraction of resources or the emission of harmful substances is never the goal of a process. The same applies to a number of economic flows: some of the flows (main products and byproducts) represent the goal of the process while other economic flows are needed to create the desired flows. In a way these other flows are undesirable because they are associated with costs. They are also undesirable further upstream or downstream in the production chain as the production or processing of these flows



also effects the environment through environmental interventions. Thus, the four categories can be subdivided into six categories: economic flows are divided into flows with a positive value and flows with a negative value. (See also Figure 2.3 and Huppes (1992).)

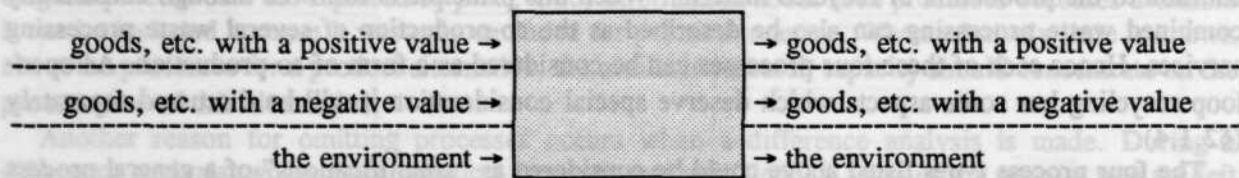


FIGURE 2.3. The economic flows in a process can be subdivided into flows with positive or negative economic values. The goal of a process is associated with the flows with a positive value.

In this way the desirability of each input and output can be determined on the basis of its value. It is therefore an obvious step to use economic value as a criterion to determine the flows to be allocated and which flows allocations should be made to. In this document economic flows with a positive value (i.e. to which allocations are made) are referred to as *main flows*. Economic flows of zero or negative value as well as flows to and from the environment are referred to as *subflows*.

This is correct when a single process is considered, here single means that there is only one main flow. This flow is the goal of the process and all subflows are allocated to it. The subflows of a multiple process are allocated to the main flows on a basis to be discussed below. However, we are still left with another problem: even when it is clear which are the main flows and the subflows, it does not mean that all the subflows have to be allocated to all main flows.

#### What is allocated to what? Causality

The principle described above is based on social causality. Subflows are allocated to main flows because the main flows are the driving force behind the process and the subflows are undesirable side-effects. This could be extended on the basis of physical and chemical principles.

The incineration of household waste can be used as an example. A wide range of different products is incinerated together. The flue gasses contain many substances which are significant in terms of environmental protection. The average composition of these emissions per tonne of waste to be incinerated has been determined empirically. However, it would not make sense to allocate a given share of all these emissions to a particular product which is incinerated. In causal terms a mercury-free product is not responsible for mercury emissions from the stack. This problem is particularly relevant to combined waste processing. There are also production processes where this problem occurs. Zinc and cadmium are emitted during the combined production of zinc and cadmium from ore. It is debatable whether zinc production can be considered to be responsible for some of the cadmium emissions.

Besides social causality (division into main flows and subflows) physical causality is important when deciding what to allocate to what. The question of how the subflows to be allocated should be allocated to the main flows arises after determining the social and physical causality.

#### How is the allocation made? General principles

A general system requirement when allocating environmental interventions is that the various processes combined should result in the process which was previously divided. Hence it is important to ensure that no environmental interventions are included twice or are not allocated\*. Therefore allocation is defined as the apportionment of all subflows to the main flows.

How should the allocation to the various main flows be made? For processes with multiple main flows a choice has to be made from at least four allocation methods:

- allocation based on the *mass proportion* of the total mass of the main flows of the process

\* These requirements must be applied to the total of all processes concerned and not just at the individual process level.

concerned;

- allocation based on the *proportion of the calorific value* of the total calorific value of the main flows of the process concerned;
- allocation based on *another physical parameter*;
- allocation based on the *proportion of economic value* of the total revenues of the main flows of the process concerned.

Allocation based on mass has so far been the most common method used (Deurloo, 1990; Driuff, 1984). Boustead & Hancock (1989) list area and number of items, in addition to mass. Driuff (1984) suggests allocation based on the economic value if the price range of the outflows per volume unit exceeds a factor of ten. This idea is developed by Huppes (1992). The use of an equivalent process is problematic as sometimes there will be several equivalent processes and sometimes there will be none.

Generally three arguments are given for selecting allocation based on mass:

- mass data is generally available or easily determined, objective and constant;
- in ideal cases the mass balance of each process will be correct;
- allocation based on mass is consistent with physical causality.

These arguments will be discussed separately.

It is indeed correct to say that mass data is almost always available and fairly constant, unlike economic data. Its objectivity however, is not as clear-cut\*, as in some cases it has to be decided whether to use the mass of a substance with or without associated water.

As it was decided to collect process data in step 2.2 of this project and to undertake the allocation at a later stage (step 2.3), the mass balance can still be used as a verification aid† for the original non-allocated multiple process data. The balance per flow, rather than the balance per process, is required to calculate the contribution of each process to the process tree in step 2.4.

Finally there is the argument of physical causality. The simple fact that calorific value or other physical parameters are used as the allocation parameter in special cases already indicates that mass is not the only relevant physical unit. When considering surface treatment operations in several products are treated simultaneously (cleaning and galvanising baths) it is obvious to use the treated area as the allocation parameter. For telephone lines which can carry multiple calls the number, or the duration of the calls would be a good parameter.

These cases demonstrate that the *function* performed by the processes may generally be regarded as the decisive parameter. In many industrial processes the function of a process is to produce a quantity of materials. In such cases the function of the process is represented by mass. In other cases the number of items, area or time a process is used will represent the function of that process.

Another question is whether the function fulfilled by a process is always clearly defined. The combined production of pharmaceuticals and fodder is an example. This process produces a much greater mass of fodder than drugs, while the economic return is largely provided by the drug production. In this case there will clearly be a major difference between allocation based on mass and allocation based on value. On the one hand it could be argued that the function of the process is based on the need to produce a quantity of fodder and pharmaceuticals. On the other hand it could be argued that the market conditions is important to the producer as it will affect the revenue and strategy.

On this basis it could be claimed that *social causality* provides the impetus to operate processes in a certain way. The ratio between the main flows can be varied within certain limits in many processes. The number of passenger cars and the number of freight cars in a train is not fixed but is selected to provide the best possible service to the market and obtain the highest possible return. Social causality is reflected in economic concepts such as price and revenue. In this light the method based on the share of economic value is the best as it is directly linked with the social causality of the process.

Furthermore it is also a method which is closely related conceptually to the first answer to the qualitative question (what is allocated to what). If it is decided to allocate flows with no value or a negative value to flows with a positive value, because flows with a positive value represent the primary

\* Huppes (1992) writes: "It seems rather odd to measure a variable with a precise but wrong measuring instrument."

† Incidentally, the calculations in an LCA do not require the input mass and the output mass of a process to be identical.



aim of the process, it would be natural to maintain this principle when determining the flow representing the most important goal of the process and to which most of the subflows should be allocated.

A disadvantage of this choice is that relatively changeable economic data about the main flow of the processes has to be available. Another disadvantage is that there is no clear economic unit. This is similar to the problem with allocation on a physical basis. A selection has to be made from gross revenue, net revenue, price after deduction of taxes and subsidies, etc. Huppes (1992) has made a detailed proposal. In essence this proposal suggests that the gross revenue is generally a good parameter for allocation.

The qualitative question about the classification of main flows and subflows has been answered with a principle of social causality and one of physical causality. If one of the arguments for using economic allocation rules is found to be based on social causality the use of physical allocation rules should also be examined.

Clearly each method has advantages and disadvantages. At present keeping the functional concept central appears to be the most acceptable. This will provide a satisfactory result in many cases. However because the functional principle cannot be directly coupled to a known quantity (mass, value, etc.) it will not always be satisfactory. In such cases two extreme situations could be studied with the aid of a sensitivity analysis. The quantitative question will be examined below for the three different situations (co-production, combined waste processing and open-loop recycling).

#### *How is the allocation made? Co-production*

To summarize: the principle of social causality provides an initial separation between main flows and subflows for co-production. Main flows are economic flows with a positive value; products, materials and services as well as the processing of waste whose value to the supplier is negative. Economic values of no value or a negative value (used products, materials and services and waste to be processed) do not provide functions to other processes and are therefore subflows which are allocated to the process concerned. The principle of physical causality is then applied for the quantitative allocation; for example, cadmium emissions are not allocated to zinc if cadmium and zinc are produced together.

We propose to use a combination of social and physical causality to determine the extent to which allocation is made. Physical allocation should be used for subflows which are clearly linked to main flows. In most cases the physical parameter used for this will be the parameter used to specify the process data. In most cases this will be mass. However there is no physical causality for most flows and the principle of economic allocation will have to be used.

The situation changes when practical aspects about the lack of required economic data are considered. As a provisional guideline for processes with more than one product mass units can be used in the calculations provided that the products can be expressed in mass units and their prices do not diverge too widely. For some processes the share in the gross revenue may be used. The average price ratio (the absolute price level is not relevant) over a longer period could be used for this. This approach is preferred for processes where the quantities involved are important on environmental grounds. For example in the basic inorganic chemical industry, the production of chlorine and caustic soda and the processing of mixed ores.

#### *How is the allocation made? Combined waste processing*

Physical causality is most important when considering combined waste processing. In many cases the chemical composition of the waste provides a good indication of the flows which should or should not be allocated. The quantitative questions about the precise allocation key can also be answered on the basis of physical causality. However, there are some difficulties in practice.

Dioxins can be emitted when carbon and chlorine are present, their formation is catalysed by copper (Kleijn & Van der Voet, 1991). This means that, in physical terms, products containing chlorine, carbon and copper are responsible for dioxin formation. However, the actual link between the presence of these substances and dioxin formation is highly dependent on the combustion conditions. For example the presence of a large fraction of wet material (putrescible waste) will depress the temperature and more dioxins will be formed.



With regard to PAH emissions it might be assumed that the presence of carbon is decisive. However, it is not clear whether or not the presence of carbon rings increases PAH emissions (Guinée *et al.*, 1988).

Nitrogen oxides emissions, however, are different again.  $\text{NO}_x$  is largely formed by the reaction of nitrogen and oxygen which are naturally present in the air. Hence the calorific value of the products incinerated will determine these emissions.

Landfilling waste is also a form of combined waste processing. The same considerations apply here as to waste incineration. It is difficult, however, to determine precisely the boundary between the product system and the environmental system. The expected emissions due to landfilled waste are discussed elsewhere (§2.2.7).

Finally, economic allocation is sometimes advocated. The price charged for processing waste varies widely, particularly for chemical waste. On the whole the price will depend on the harmfulness of the waste: processing highly polluted waste is much more expensive than processing less harmful chemical wastes. However, this does not provide information about its harmfulness before and after processing as a treatment plant has to meet statutory requirements, hence highly polluted waste has to be treated more intensively or for a longer time. A more intensive treatment can of course be more harmful to the environment, this will be revealed when the entire process operations are included in the life cycle.

To conclude, it could be claimed that physical causality is dominant with respect to the creation of certain emissions in combined waste processing. The exact causalities are not clear for all substances emitted. Some emissions depend on the chemical composition of the products while other emissions depend on the calorific value. Furthermore, there are substances which promote emissions. Essentially there are two approaches to the problem:

- from the input angle: a cadmium-containing product can, in principle, emit cadmium or cadmium compounds when incinerated;
- from the output angle:  $\text{NO}_x$  emissions due to waste incineration are caused by one or more of the incinerated products.

The problem is that these two approaches have to be coordinated. A satisfactory solution for the allocation problem will have to be determined in each individual case. In principle this is feasible (Petersen, 1992). It is not clear, however, how a satisfactory solution can be found within a reasonable period when carrying out practical studies. Therefore allocation in relation to combined waste processing requires further study.

#### *How is the allocation made? Open-loop recycling*

Open-loop recycling occurs when materials, components or products from one product system are collected, cleaned or upgraded for use in another product system. The resulting allocation problems include:

- which product system should the collection and upgrading process be allocated to?
- should the mining of resources (in terms of a process) and the extraction (in terms of depletion) be allocated in full to the product which the original resources were used for?
- should the processing of the eventual final waste be allocated in full to the product in which the material was last used?
- should the deterioration of a material in use be allocated, if so how?
- to what extent should contamination of the secondary material be allocated to the receiving product?
- should we distinguish between recycling, reuse and useful application of materials?

These questions show that recycling should be considered separately and that the solutions chosen for co-production and combined waste processing do not always apply in full. For these reasons recycling is discussed in a separate section

#### **2.1.4 Recycling and reuse**

Two situations are relevant when considering recycling\*:

\* In the remainder of this section we will use the term "recycling" for both recycling and reuse. This is because a definite

- closed-loop recycling;
- open-loop recycling.

Closed-loop recycling is the upgrading or reuse of materials, components or products for the same product system. Examples include bottles which are collected via a return system (possibly by means of a deposit), cleaned and reused, as well as old glass collected in the bottle banks which is molten and reused. These reuse processes may result in a reduction in the use of raw materials and auxiliary materials and in the production of waste to be processed. The subflows associated with reuse processes are an undesirable side effect. Both the advantages and disadvantages remain in the product system as both the flow (waste) to be upgraded as well as the upgraded flow are generated by the product system\*.

The processes in the chain are linked when a material or products in the life cycle of a product are recycled. Recycling processes enable savings in that material or those products. The consequences of recycling in a life cycle include aspects which both reduce and increase environmental interventions. Hence, recycling has advantages as well as disadvantages.

Generally the advantages (i.e. reduction in environmental interventions) will be self-evident:

- recycling reduces the waste processing of disposed products and materials thus reducing emissions, etc.;
- recycling reduces the use of resources and thus some of the environmental interventions.

In the final assessment the potential disadvantages of recycling (increase in environmental interventions) must also be considered. Recycling will generally require certain upgrading processes such as transport, washing and distillation. Such additional processes generally result in further environmental interventions by the product chain.

In an LCA recycling should be assessed on the basis of the actual situation as the allocation of recycling requires a consideration of the advantages and disadvantages which will always largely depend on the actual situation. It is possible to include some general characteristics and constraints for recycling in the methodological framework. The recycling of aluminium will be described below as an example of recycling in a product chain.

#### *Open-loop recycling and closed-loop recycling*

In an LCA the product chain is assessed after the delineation of the system and the inventory analysis based on the aggregation of the environmental interventions of all the individual processes. The product chain will be fairly simple in an LCA if there is no recycling of products or materials. This follows from the diagram of the product chain in Figure 2.4. Without recycling there would be no feedback between the processes in the product chain.

Two forms of recycling have to be distinguished in the system delineation. These are closed-loop recycling and open-loop recycling. In closed-loop recycling (described above) the discarded product or material is reused in the original product or material or a similar one. Thus, recycling is limited to the same product chain. Therefore there are no delineation problems. However, in open-loop recycling the discarded product or material is reused in another product chain. Here two or more product chains are linked.

Closed-loop recycling is described in Figure 2.5. Please note that the recycling process step generally includes several different processes such as transport, cleaning, upgrading and shredding. For convenience, these processes have been included in a single block in the figure. It is important to note that closed-loop recycling is fully included *in the product system* of the original product chain. When a product chain includes closed-loop recycling the changes in the environmental interventions resulting from a reduction in the use of resources, etc. and a reduction in the processing of final waste as well as the additional interventions due to the recycling process all have to be considered.

The extent to which the recycling in the product system affects the quantification of all the products

distinction is not necessary for LCA: material produced by an economic process is made suitable for use as a material in an economic process by one or more recycling processes (collection, cleaning, upgrading, etc.).

\* An LCA can be used to investigate for each product system whether the disadvantages of recycling (use of raw materials, emissions) outweigh the benefits (Boustead, 1990). A scenario study could be used to determine the optimum recycling level.

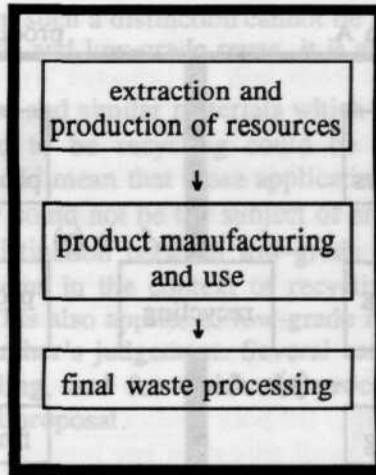


FIGURE 2.4. Process tree diagram of a product system without recycling.

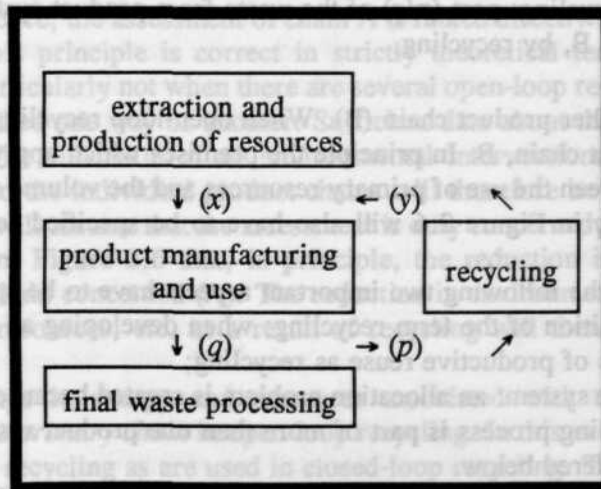


FIGURE 2.5. Closed-loop recycling: part  $(p/q)$  of the waste is partly reused  $(x/y)$  through a recycling process.

affected should of course be specified when closed-loop recycling is used. In other words,  $x$  and  $y$  in Figure 2.4 must be specified, where  $x/y$  is the ratio between the use of primary and reused material. Additionally  $p$  and  $q$  have to be specified:  $p/q$  is the ratio between the waste to be processed and that for recycling. The above ratios  $x/y$  and  $p/q$  must be specified for each situation separately. Some examples follow below:

- For the production of newsprint for example we could assume 80% recycling, i.e.  $x/y = 20/80$ . This ratio is determined by the requirements of newsprint. Copier paper, however, is subject to completely different requirements and has a different  $x/y$  ratio.
- For waste glass packaging in domestic refuse it could be assumed that 90% is collected separately and reused:  $p/q = 10/90$ . This ratio would then depend on the actual effects of the separate collection.

Hence, both ratios  $(x/y$  and  $p/q)$  are parameters in the LCA method and should be specified separately. In practice this information is included in tables. Such underlying information for the LCA of a production system will generally be based on specific practical situations such as technical constraints, requirements imposed on reused materials, response to waste collection, etc. or expectations about these matters.

Figure 2.6 illustrates open-loop recycling. The discarded product and/or material from a product



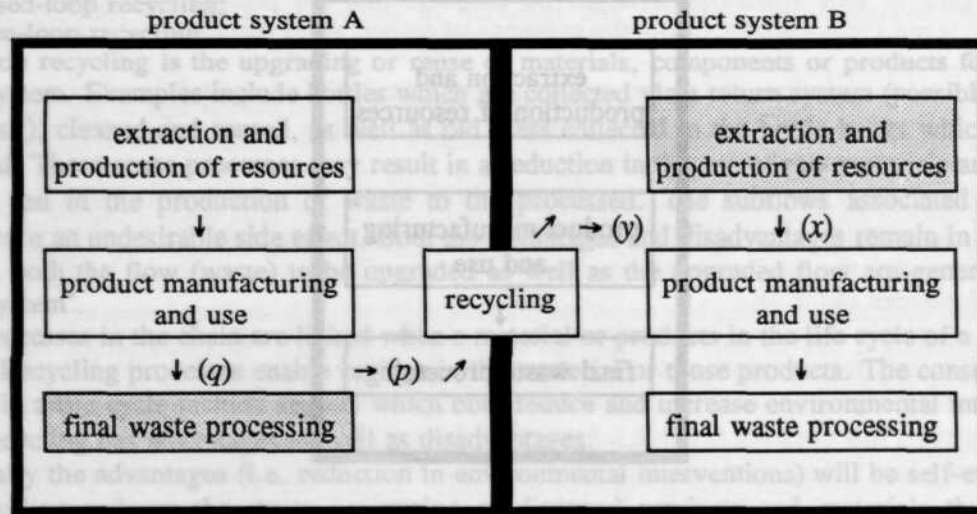


FIGURE 2.6. Open-loop recycling: part ( $p/q$ ) of the waste from product system A is partly used ( $x/y$ ) in another product system, B, by recycling.

chain (A) is reused in another product chain (B). When open-loop recycling is used product chain A is linked to another product chain, B. In principle the premises which apply to closed-loop recycling also apply to the ratio between the use of primary resources and the volume of reused material. Hence the parameters  $x/y$  and  $p/q$  in Figure 2.6 will also have to be specified separately when open-loop recycling is used.

In open-loop recycling the following two important aspects have to be specified:

- a more precise definition of the term recycling: when developing an LCA it may be useful not to consider all forms of productive reuse as recycling;
- the delineation of the system: an allocation problem is created because two product systems are linked and the recycling process is part of more than one product system.

Both aspects will be considered below.

Clearly the diagrams in Figure 2.5 and Figure 2.6 are relatively simple. In practice there are many combinations of open-loop recycling and closed-loop recycling (e.g. co-product recycling) and cascades of open-loop recycling are also common.

#### *A narrower definition of open-loop recycling*

In practice recycling is a rather broad term. Recycling includes not just the reuse of materials but also the reuse of products and residues. The useful application of waste (secondary and tertiary recycling, e.g. incineration and composting) is also often called recycling. The need for a strict definition of the term recycling is demonstrated by the following example.

Let us assume that product system A is a mercury switch (a component of a refrigerator or heater). In the waste state the switch is included in domestic refuse and leads to relatively large mercury pollution of the flue gas scrubbing residue. As a result the waste has to be landfilled as chemical waste with all its associated environmental interventions unless another application can be found. Let us also assume that, under strict conditions, the flue gas residue could be used as an alternative filler. Although flue gas residue is not a particularly attractive replacement for the original filler this is the only alternative to landfilling it as chemical waste. According to the broad definition of recycling this example results in open-loop recycling. Depending on the allocation method used important environmental aspects may be left out of the product system under consideration when this interpretation of open-loop recycling is used in an LCA of a mercury switch. This is because the environmental interventions associated with the waste stage of the secondary product (i.e. the application as a filler) are not included in the investigation of the mercury switch.

This example suggests that perhaps open-loop recycling should only be applied to the high-grade

reuse of waste materials. At present such a distinction cannot be made in practice\*. A clear distinction cannot be made between high-grade and low-grade reuse, it is also likely that there is a sliding scale rather than a definite distinction.

A list of applications of residues and similar materials which are considered low-grade and where the application is not considered to be recycling could be drawn up by agreement (possibly internationally). However, this would mean that those applications could no longer be interpreted as separate product systems and they could not be the subject of an individual LCA. At present it is not possible to make such a clear distinction between low-grade and high-grade applications. When discussing the delineation of systems in the context of recycling we mentioned that open ends in process trees should be flagged. This also applies to low-grade reuse where further indications could be given on the basis of the researcher's judgement. Several categories of useful applications which need not be considered as recycling, and for which the process tree need not be extended, are identified in the guide as an initial proposal.

#### *System delineation and open-loop recycling*

At present there is no agreement at international level about system delineation when open-loop recycling is used. According to Hunt *et. al* (1974) both product chains are firmly linked when open-loop recycling is used. Hence, the assessment of chain A is linked directly to the assessment of another product, B. Although this principle is correct in strictly theoretical terms generally it cannot be developed in practice, particularly not when there are several open-loop recycling cascades. In the end all product chains are linked one way or another. Sufficient data about the product chains concerned is often not available. The allocation of all environmental interventions (increases and decreases) influenced by recycling to the individual product chains will therefore be difficult.

For these reasons we will assume that the assessment is only concerned with a single product chain. It can be concluded from Figure 2.6 that, in principle, the reduction in waste processing will be included in the product chain concerned (A). The reduction in environmental interventions due to the reduction in the use of resources, etc. as a result of recycling will then be included in subsequent product chains (B).

In practice the savings in energy and resources associated with the secondary material are sometimes included *in the primary chain* in open-loop recycling. In this event the same allocation rules are applied to open-loop recycling as are used in closed-loop recycling. This is incompatible with the above allocation principle for open-loop recycling. When the consequences are examined more closely it appears that it would be incorrect to deviate from this allocation principle. Strictly speaking, all aspects of the primary extraction of resources, etc. should be allocated to the last application of a material if this principle is set aside and the above approach is adopted which sometimes occurs in practice. For example, when plastic is used all the environmental interventions relating to the extraction of raw materials and polymer synthesis would be allocated in full to the last application of the material, e.g. a verge post. Thus this method is hardly realistic. In this way the last product would bear the full burden. This would definitely lead to incorrect considerations with respect to recycling.

In one marginal case it can be *proven* that the secondary material which is not reused in the primary production chain studied results in primary material savings in another product chain. In that case the other, secondary, chain is essentially the primary chain for that material. In a way this is a case of closed-loop recycling. In marginal situations of this kind there are arguments for including the savings in energy and resources in the primary chain. When this is done in an LCA the non-standard allocation method and the premises used should be stated fully.

This form of system delineation has a restriction: this method clearly makes it impossible to provide a *complete* overview of the consequences of the recycling based on the inventory analysis of one product system. When open-loop recycling is used the recycling process step is situated *between* the delineations of the two product chains (see Figure 2.6). The environmental interventions due to the

\* The only criterion could be whether the waste has a positive or negative value. In practice this is unclear as waste is sometimes reused within a company in which case there is no market price. In other cases waste may have an artificial positive price when government subsidises the cost of collection.



processes required for recycling will still have to be allocated to the two product systems. This aspect of the allocation system has yet to be developed. Depending on the allocation method used product system A or product system B or both will be affected by the disadvantages (environmental interventions and economic subflows) of the intermediate upgrading steps.

The literature provides several options for allocating of recycling step to both product systems:

- mass based allocation with a ratio of 50/50 (Fava *et al.*, 1991);
- an allocation key based on the ratio of the economic values of the waste to be processed and the recycled product (Huppel, 1992);
- full allocation to product system A or B.

The first method (50/50) is arbitrary as the allocation key cannot be supported. If product system B can be described, which is not always possible, the method is clear. However one of the disadvantages of this method is that does not provide any encouragement to promote recycling. When full recycling is not possible, e.g. due to the poor quality of the waste, this should really be attributed to the causal product chain A. The overall allocation key remains 50/50, irrespective of the level of recycling. Hence, this allocation method provides no incentive to product chain A to improve waste quality and increase recycling.

Theoretically, the economic allocation method could remove these constraints. Let us assume that using this method a reasonable waste quality would result in a 50/50 distribution. If only part of the waste could be reused due to poor quality this will reduce the economic value of the waste and process chain A will receive a larger share of the disadvantages of recycling. However the economic price does not always provide an accurate reflection of the quality of the material. The price of the material also depends on the options for functional reuse which vary. Hence allocation will not be uniform in practice.

Clearly complete allocation to one of the two product systems is very easy. However this would be arbitrary and often provide no incentive to improve the quality of the waste and to increase recycling.

Various other allocation methods are suggested in the literature. Lindeijer & Fraanje (1992) also allocate part of the environmental interventions associated with the extraction and production of primary resources to the cascade of subsequent product chains. In practice these and other allocation methods are not consistent as the cascade of product system is not constant. Furthermore when this method is used part of the environmental interventions of the waste processing in the following cascades should also be allocated to the primary product chain.

A clear and consistent method of system delineation is required for a practical approach to dealing with open-loop recycling. Hence the third method is preferred. However such a system would not provide any incentive to increase recycling. Strictly speaking this corresponds with the earlier restriction resulting from the delineation of the system. To remove this restriction to some extent an LCA should always indicate when a transfer to another product chain has been made via open-loop recycling. This means that the open ends in a process tree should be flagged. The next process tree should be developed *completely* if the study includes an assessment of all the effects of recycling. As this will generally not be practical at least an indication of these environmental interventions should be given.

#### *An example of open-loop recycling: aluminium*

Figure 2.7 outlines the life cycle of aluminium. When aluminium is used a clear distinction can be made between primary aluminium and secondary (reused) aluminium. It appears that, for aluminium, the above diagrams describing open and closed-loop recycling (Figure 2.6 and Figure 2.5) are greatly simplified. Internally both the primary and the secondary aluminium chains contain closed-loop recycling. An investigation of aluminium window frames (Lindeijer & Fraanje, 1992) indicated that there are several cascades of open-loop recycling for secondary aluminium.

When an LCA is undertaken this complex situation should be borne in mind when considering the opportunities for recycling (determining the ratios  $x/y$  and  $p/q$  in Figure 2.6 and Figure 2.5). The opportunities for recycling are largely determined by the requirements concerning the material composition and the purity of the scrap aluminium. The extent of recycling will have to be determined and stated in each case (see also BUWAL, 1991).



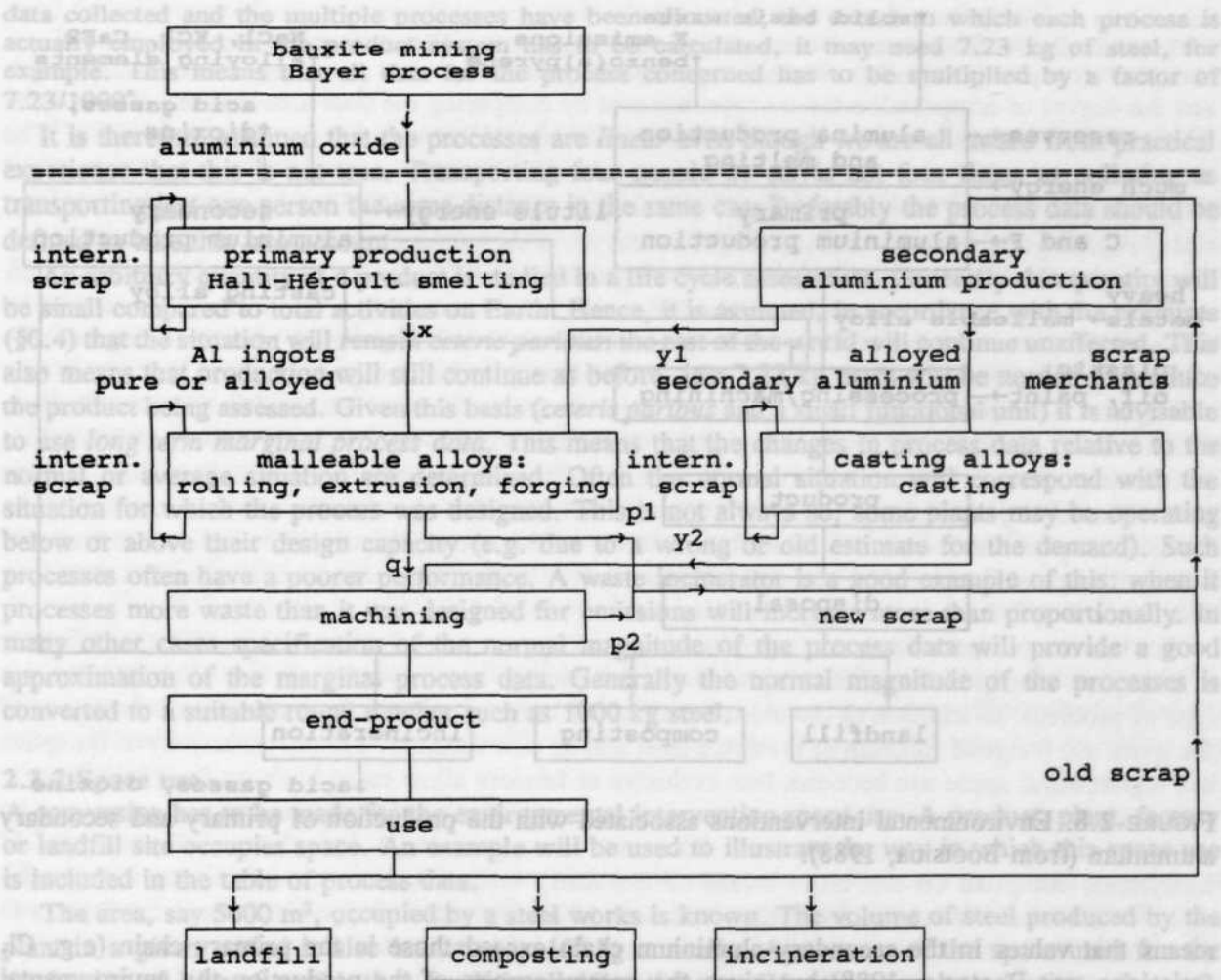


FIGURE 2.7. The life cycle of aluminium (from Bootsma, 1988).

Figure 2.8 shows that there is a range of environmental interventions in both the primary and the secondary aluminium chains. This also applies to the production of secondary aluminium. In essence this process corresponds to the recycling process step in the diagram in Figure 2.6: the scrap aluminium is collected, cleaned and heated before melting it to secondary aluminium. In accordance with the above considerations the environmental interventions in the intermediary processes should be allocated either to the primary chain or the secondary chain. The following remarks apply to this:

- Some aluminium processing methods are only suitable for primary aluminium. This does not apply to all techniques (e.g. casting alloys). Although the opportunities for the use of secondary aluminium are limited there clearly is recycling: the reuse of aluminium is not low-grade reuse.
- The production of primary aluminium requires a considerable amount of energy. The energy (i.e. electricity) generation is accompanied by considerable environmental interventions. These are fully allocated to the primary aluminium chain, not to the secondary chain.
- The environmental interventions associated with aluminium waste (final waste) are unimportant compared with the environmental interventions associated with its extraction and production. The environmental interventions of aluminium waste are fully allocated to the secondary aluminium chain.
- The environmental interventions due to the upgrading of scrap aluminium to secondary aluminium are limited by comparison with the interventions for its extraction and production and also depend largely on the requirements of the product application on the secondary aluminium.

In view of these considerations the environmental interventions due to the upgrading of secondary aluminium are generally allocated in full to the secondary aluminium chain. For some emissions this

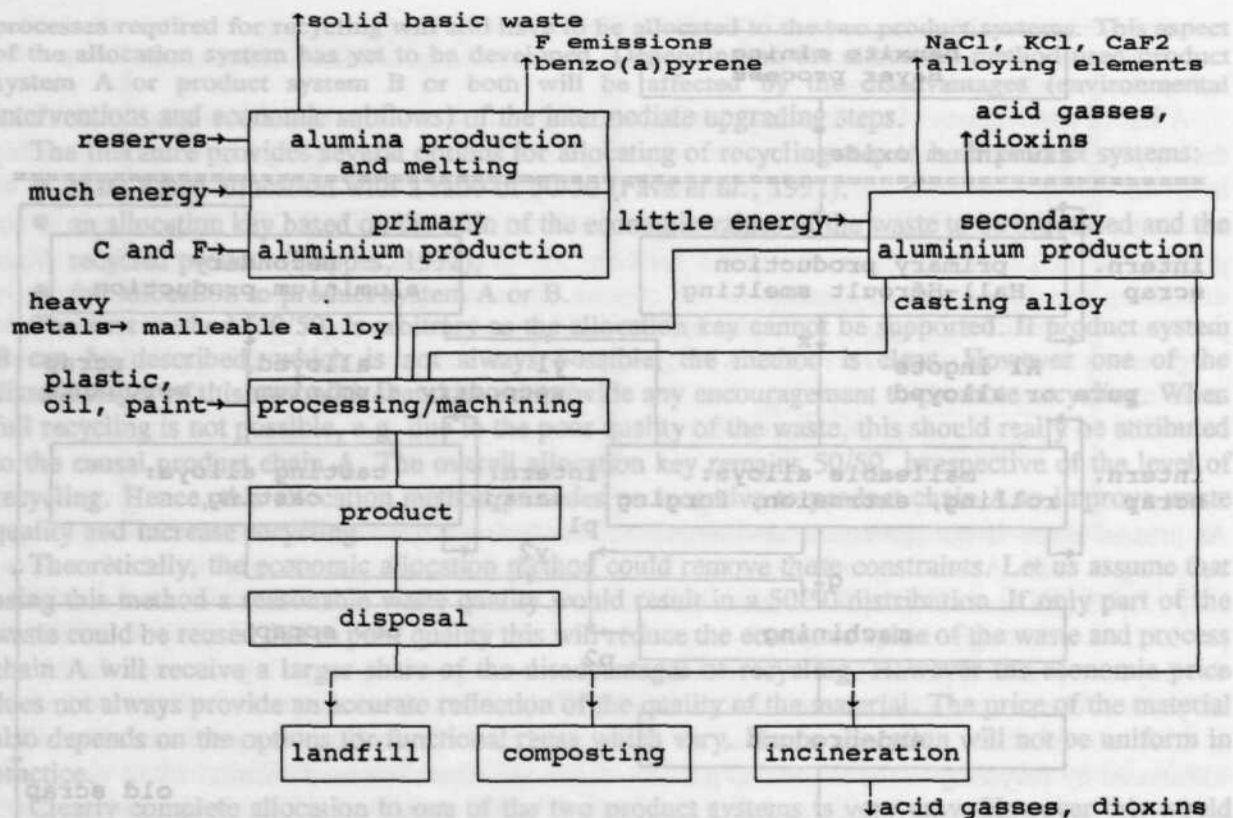


FIGURE 2.8. Environmental interventions associated with the production of primary and secondary aluminium (from Bootsma, 1988).

means that values in the secondary aluminium chain exceed those in the primary chain (e.g.  $Cl_2$  emissions, see Bootsma, 1988) but given the energy aspects of the production the environmental interventions of the primary chain in total exceed those of the secondary chain.

The above method of allocation is therefore simple and appears to be fair in relation to both the primary and the secondary aluminium chain. However, this is not to say that the same allocation to the primary and secondary chains will apply for any material. For example, for another material the environmental interventions due to waste processing might be important and the environmental interventions associated with upgrading to secondary material might be comparable to those for the extraction and manufacturing of the primary material. In this event it would be more natural to allocate the environmental interventions of the intermediate step to the primary chain only.

## 2.2 The process data

Economic processes are associated with many environmental interventions: the extraction of resources and the emission of substances as well as some less obvious interventions which may have various effects. The SI system can be used for all emissions and extractions: many emissions and resources can be expressed in kg or its derivatives (mg,  $\mu$ g, etc.). Other energy data is expressed in kWh, MJ, etc.

Noise is an environmental intervention which requires special treatment. Normally noise is expressed in dB, however it appears that this unit is not suitable in LCAs. Noise and some other special process parameters will be discussed below.

### 2.2.1 The linearity of the processes

Process data is generally specified in certain units: the process characteristics of a process to produce 1000 kg of steel can be found in the literature. Once the process tree has been drawn up, the process

data collected and the multiple processes have been allocated, the extent to which each process is actually employed in the product system has to be calculated, it may need 7.23 kg of steel, for example. This means that all data for the process concerned has to be multiplied by a factor of 7.23/1000\*.

It is therefore assumed that the processes are *linear* even though we are all aware from practical experience that this is not true. Transporting four people by car is not four times as polluting as transporting just one person the same distance in the same car. Preferably the process data should be defined to take this into account.

An arbitrary quantity of a product is studied in a life cycle assessment. Generally this quantity will be small compared to total activities on Earth. Hence, it is assumed, in accordance with the premises (§0.4) that the situation will remain *ceteris paribus*: the rest of the world will continue unaffected. This also means that production will still continue as before, just 7.23 kg extra will be needed to produce the product being assessed. Given this basis (*ceteris paribus* and a small functional unit) it is advisable to use *long term marginal process data*. This means that the changes in process data relative to the normal or average situation are determined. Often the normal situation will correspond with the situation for which the process was designed. This is not always so, some plants may be operating below or above their design capacity (e.g. due to a wrong or old estimate for the demand). Such processes often have a poorer performance. A waste incinerator is a good example of this: when it processes more waste than it was designed for emissions will increase more than proportionally. In many other cases specification of the normal magnitude of the process data will provide a good approximation of the marginal process data. Generally the normal magnitude of the processes is converted to a suitable round number such as 1000 kg steel.

### 2.2.2 Space use

A conversion has to be made for the environmental intervention space use. A product, plant, factory or landfill site occupies space. An example will be used to illustrate the way in which this space use is included in the table of process data.

The area, say 5000 m<sup>2</sup>, occupied by a steel works is known. The volume of steel produced by the plant in a given period is also known, say 10<sup>6</sup> kg annually. A measure of the space used for the production of 1 kg of steel is then found by dividing the area  $O$  by the annual production volume  $Q$ . This parameter is the *specific space use*  $\rho$ , in the example above its unit is m<sup>2</sup>·yr·kg<sup>-1</sup>:

$$\rho = \frac{O}{Q} \quad (2.1)$$

A process specification includes the quantity of product manufactured, e.g. 1000 kg. The associated *space use*  $R$  (unit: m<sup>2</sup>·yr) is determined by multiplying the specific space use  $\rho$  by the quantity produced  $q$ :

$$R = q \times \rho \quad (2.2)$$

The space use in the example given above is  $1000 \times 5000/10^6 = 5$  m<sup>2</sup>·yr.

Other processes will have a different typical unit of the product manufactured. The production of a power station is specified in MJ and the specific space use  $\rho$  is specified in m<sup>2</sup>·yr·MJ<sup>-1</sup>. This does not affect the parameter we are actually interested in: the space use  $R$ .

Strictly speaking, space use is determined not by the annual production but by the production during the period that a plant occupies space divided by the time the space is occupied. In other words, this should include the nonproductive time of a factory site required for construction and demolition. For most processes the nonproductive period is negligible by comparison with the productive period. A nuclear power station provides an example of an exception as its decommissioning may take a very long time. However, the nonproductive time is not yet fully understood. A factory site which requires decontamination after the demolition of the factory is another difficult example. Information about this is generally unavailable while the plant is operating,

\* This is only possible if the process data are specified in a form which permits linear operations. This effects certain parameters (space use, noise), which are discussed elsewhere.



hence it is normally excluded.

In principle the same method can be used when waste is landfilled or stored. In this case the area occupied by a product in  $m^2$  is relevant. Appropriate parameters such as the maximum landfill depth and the degree of compression can be approximated by comparing the total area with the total volume of waste deposited. An attempt could be made to distinguish between levels of compressibility. The time waste is landfilled is generally limited as a landfill site will eventually be grassed over and acquire a new use\*, for example, as a golf course. An end time is not defined for extremely long term storage (storage of batteries and jarosite; Chernobyl). This would mean that there is a second form of space use which is independent of time. This parameter will be referred to as *final space use*; its unit is  $m^2$ . Normally this parameter will be little used and an extremely conservative estimate will be made of the time. For batteries, for example, which are now being stored while awaiting suitable processing techniques a period of 100 years could be used as the storage time.

Physical damage to ecosystems is one of the environmental effects included in the classification. This includes all direct effects on ecosystems which are not due to the use of resources, hazardous substance emissions, noise or radiation. Damage to ecosystems is a regional or local problem, similar to the issue of drying out (due to the falling water table). In a study of window frames the use of the resource tropical hardwood was classified in terms of the use of resources (in kg) and in terms of damaged tropical forest (in hectares) (Lindeijer *et al.*, 1990). This example illustrates that there is a link between space use and damage to ecosystems.

For this reason space use is employed as the inventory parameter for this problem. The significance of the space use, in terms of the displacement and damage to ecosystems, is also evaluated for this type of problem. In addition to the use of space as such this provides information about the nature of the space use (original function of the space used and its new function) and the exclusivity of the space use (agricultural space use becomes less exclusive as farmers allow more birds on their land).

An inventory analysis of damage to ecosystems requires that different ecosystem types can be identified. This could be based on the flora and fauna present. A list of the different types of ecosystems compiled on this basis would be extremely long. Furthermore a factor to indicate the relative values of different ecosystems would have to be developed. Suggestions about the aspects to be included in such a factor have been made elsewhere (Lindeijer & Fraanje, 1992; Fraanje *et al.*, 1990). Such a factor could then be multiplied by the size of the area in hectares, which is damaged by an economic process. Clearly the classification of different ecosystems and the development of valuation factors would require a major effort, if it would even be feasible. For these reasons a different approach will be suggested below.

The ecosystem categories defined in an IUCN report (IUCN/WWF/UNEP, 1991) can be used for a coarser estimate of the significance of the space use in terms of dislocation and damage. This approach was promoted by Frischknecht *et al.* (1992) and Finnveden & Lindfors (1992) and is also used here. These categories integrate the two aspects (type and exclusivity) of the significance of the space use. Five ecosystem types are identified:

- natural systems;
- modified systems;
- cultivated systems;
- built systems;
- degraded systems.

The original definitions (IUCN/WWF/UNEP 1991) are given below.

Type I, natural systems are "Ecosystems where since the industrial revolution (1750) human impact (a) has been no greater than that of other native species, and (b) has not affected the ecosystem's structure. Climate change is excluded from the definition, because human-caused climate change is likely to affect all ecosystems and eliminate all natural ecosystems as defined here".

Type II, modified systems are "Ecosystems where human impact is greater than that of other species, but whose structural components are not cultivated. Most of the planet is now modified, including land and sea areas usually considered 'natural'. For example, naturally regenerating forest

\* The aim of multifunctional use is a matter of debate.

used for timber production; naturally regenerating land used for livestock production".

Type III, cultivated systems are "Ecosystems where human impact is greater than that of any other species, and most of whose structural components are cultivated: e.g. farmland, sown pasture, plantations, aquaculture ponds".

Type IV, built systems are "Ecosystems dominated by buildings, roads, railways, airports, docks, dams, mines, and other human structures".

Type V, degraded systems are "Ecosystems whose diversity, productivity and habitability have been substantially reduced. Degraded land ecosystems are characterized by loss of vegetation and soil. Degraded aquatic ecosystems are often characterized by polluted water that can be tolerated by few species".

Each use of space could be defined as a change from one category to another. For example, the construction of an airfield in a polder landscape results in a change from III to IV\*. This provides a further parameter for the space use of a plant. The number of square metres required for a given type of space use is relevant, (similar to the specification of the number of kg and the substance for an emission). Ten types of space use can be distinguished: I → II, I → III, I → IV, I → V, II → III, II → IV, II → V, III → IV, III → V en IV → V. Another method uses the definition of four standard changes in space use: I → II, II → III, III → IV en IV → V. According to this method a change from category I to category III of 10 m<sup>2</sup>·s would be included twice as 10 m<sup>2</sup>·s: in the change I → II and in the change II → III. The first method, in which ten types of change are used, appears to be better as it provides more information.

At present there is no detailed classification of all possible types of space use in these categories. The IUCN has made the start described above. It is likely that this will not be sufficiently detailed for LCAS. Based on experience obtained with actual assessments it might be possible to refine the classification.

### 2.2.3 Noise

The environmental intervention noise is associated with a problem which can be solved in a similar way as the time dimension associated with space use. A process produces a certain amount of noise, specified in dB, which is to be apportioned linearly to a unit of material or product produced. The square of the sound pressure, a unit of sound energy, provides a linear parameter for the amount of sound (Pulles, 1992). This parameter, with the unit Pa<sup>2</sup>, is referred to as the *sound energy*<sup>†</sup>  $U$  and is derived from the sound pressure level  $p$  (in dB) with:

$$U = U_0 \times 10^{p/10} \quad (2.3)$$

in which  $U_0$  is a constant ( $4 \cdot 10^{-10}$  Pa<sup>2</sup>) related to the threshold of hearing (0 dB). Unlike the sound pressure, level the sound energy can be apportioned linearly to a product.

When space use (§2.2.2) was considered, the time during which an area was occupied for a given product was relevant in addition to the area taken up. The same applies to sound: the time during which noise is generated is relevant, as well as the sound pressure level. In the same way as for space use the *specific noise production*  $\gamma$  (in Pa<sup>2</sup>·s·kg<sup>-1</sup>) is determined by dividing by the annual production  $Q$  (in kg):

$$\gamma = \frac{U}{Q} \quad (2.4)$$

while the *noise production*  $G$ , in Pa<sup>2</sup>·s, is determined with:

$$G = q \times \gamma \quad (2.5)$$

where  $q$  represents the required quantity.

In the process data noise production has the same status as space use and the more traditional

\* A change in the reverse direction is also possible. For example a factory site may be cleared up to restore a polder landscape (IV to III). This will then be assessed as a negative change from III to IV and will be subtracted rather than added to the effect score for physical damage to ecosystems.

† The dimension Pa<sup>2</sup> corresponds with the square of the energy density ( $(J \cdot m^{-3})^2$ ). Hence the term energy is not quite correct.



parameters (extraction, emission and manufacture of products). Again the specific noise production may have a different unit, e.g.  $\text{Pa}^2 \cdot \text{s} \cdot \text{MJ}^{-1}$ , in some processes but this will not affect the eventual intervention (noise production) its unit remains  $\text{Pa}^2 \cdot \text{s}$ .

#### 2.2.4 Accidents and disasters; victims

The dangers of economic processes can be divided rather arbitrarily into accidents and disasters. The main difference between them is the scale and frequency: disasters are incidents which are so serious that they should only rarely occur. As the distinction is unclear it is proposed to apportion all disasters (based on risk assessments) and all accidents (based on statistics) to a product.

Accidents and disasters may be associated with environmental interventions: toxic substances emissions, radiation, noise and odour. The normal environmental interventions can be expanded with accidental and disastrous environmental interventions using the average or estimate frequency of these incidents.

A fifth type of environmental intervention is important in addition to the disastrous environmental interventions: victims. This includes fatalities and injuries which may be caused as a direct result of an accident or disaster. This definition excludes indirect victims, animals and intentional victims. Direct fatalities are e.g. due to a fire, explosion or traffic accident. Naturally the distinction between direct and indirect is unclear: someone killed by flames is a victim, but what about somebody asphyxiated by smoke? Nevertheless, this definition appears to be suitable to include the number of potential victims in an LCA. In some cases the inclusion of animal victims could be defended, e.g. birds which hit electrical transmission wires, badgers killed by traffic or birds killed by an oil tanker disaster. This will often be impossible due to the lack of data. When animal victims are included it is likely that only vertebrates will be considered: nobody would suggest the inclusion of flies splattered by car headlights. Only unintentional victims are included. Hence, if animals were considered, hunting and livestock management would not be included.

According to current legislation (EC, 1982; VROM, 1988b) a risk assessment of the consequences of possible disasters should include an estimate of the likelihood of such a disaster and an estimate of the potential number of human victims as a result of such a disaster. The number of victims in particular can only be estimated if the local conditions, such as the location of residential areas in relation to a given process, are known (Vermeire *et al.*, 1992). Victims could be apportioned to a process in generic terms, e.g. by multiplying the chance of an accident  $p$  (in  $\text{s}^{-1}$ ) by the potential number of victims per accident  $N$  (dimensionless) and by dividing the result by the annual production  $Q$  (e.g.  $\text{kg} \cdot \text{s}^{-1}$ ):

$$\sigma = \frac{p \times N}{Q} \quad (2.6)$$

The parameter  $\sigma$  (specific victims in  $\text{kg}^{-1}$ ) could be multiplied by the quantity produced of the product  $q$  (in  $\text{kg}$ ) to arrive at the number of environmental intervention victims (dimensionless):

$$S = q \times \sigma \quad (2.7)$$

The distinction between relatively infrequent disasters with major consequences and relatively frequent accidents with marginal consequences is not relevant for this apportionment. The number and type of victims (slightly injured, seriously injured and fatalities) can be extrapolated from statistics on similar disasters or may have to be estimated. Such data should be collected during the inventory analysis. Alternatively the potential victims of disasters are apportioned in general terms to each process, these could be divided in three categories:

- slightly injured;
- seriously injured;
- fatalities.

In most risk assessments only the number of human fatalities is estimated. Hence, this type of problem will be limited to this category of victim for the time being.



### 2.2.5 Radiation

The release of ionising radiation is an important problem. The corresponding section on the classification of radiation suggests that it will continue to be difficult to deal with radiation. The inventory analysis will have to provide information about the release of radiation if this aspect is to be included in the classification. This is data about the radiation intensity (in Bq), the energy of the radiation (in eV) and the type of radiation ( $\alpha$ ,  $\beta$ , etc.). It may be assumed that under normal conditions the absorption and health effects of artificial radiation\* will be negligible†. Many factors such as the use of protective apparel affect the absorption of radiation, particularly at work. Furthermore there is radiation due to past accidents and nuclear weapons tests. It is doubtful that risk assessments about the release of radiation in accidents is useful to an LCA.

### 2.2.6 Light

Light interferes with the sleep of certain animals. This problem is relevant on an extremely localized scale in greenhouse horticulture. The total influence on the environment is relatively small (Bertels, 1992). Light is not included in the standard classification model and need not be investigated in the inventory analysis. It can be included if necessary in the same way as space use (§2.2.2) and noise (§2.2.3) where the time during which the intervention occurs (i.e. light is emitted) and the magnitude of the intervention (light intensity in lumen) are relevant. This results in an intervention with the unit  $\text{lm}\cdot\text{s}$ . It is questionable whether special light sources which affect human health (e.g. welding equipment which affects sight) should or could be included under the environmental problem of light.

### 2.2.7 Emissions from final waste

Landfilling waste leads to more than final or long-term space use. Although many other LCA methods consider final waste as an environmental effect (Fava *et al.*, 1991; ILV, 1991), final waste is not considered an environmental effect in this report (see also Guinée *et al.*, 1990; Guinée, 1992a; Finnveden, 1992). Like waste incineration, landfilling waste is an economic process. Its economic inputs include waste and it results in emissions and space use. A major difference between landfilling and other economic processes is its lagging effect. In industrial activities the process concerns an activity which persists for a limited time. The production of materials and products as well as the provision of services occupy space and emit harmful substances while the activities exist. In contrast landfilling is associated with little if any activity: the waste is stored and will continue to release substances and occupy space. This has some special consequences.

Every landfill site causes emissions to the atmosphere, soil, groundwater and surface water. The nature and scale of the annual emissions every year depend on the waste composition and the quality of the storage. For example, let us consider a landfilled cadmium-containing product. The cadmium will be released into the environment slowly but persistently due to leaching. Most or all of the cadmium will be introduced into the environment after a sufficient period of time. The cadmium may be transported by the percolation water, irrespective of the quality of the landfill site. The quality of the storage (cover, foil base) only affects the rate at which a substance is introduced into the environment, i.e. the time that will pass before, say, 99% has left the landfill site.

A landfill site may be considered to be well managed (and will meet the Dutch IBC criteria‡) when the total annual emissions result in a concentration which does not exceed a given maximum such as the natural background concentration. The requirements with regard to maximum emissions apply to the entire landfill site not just to the waste product dumped there. It would therefore be unreasonable to apply these criteria to the cadmium emissions of the landfilled product. These criteria are more

\* Sources such as cosmic radiation, natural background radiation, etc. are excluded.

† Naturally radiation absorbed as a result of e.g. X-ray diagnosis is not included. The reasons for this are the same as those for excluding the potentially toxic effects of taking drugs. There are some interesting borderline cases: it is debatable whether the effects of pesticides in food on human health should be regarded as an *environmental problem*. The pesticide did not actually enter the environment, it remained within the product system. However, as human toxicity is classified as an environmental problem the answer to this question may be affirmative.

‡ Described in the Bulletin of Acts, Orders and Decrees of May 1990. This description will be amended.

relevant in the context of issuing a licence or preparing an environmental impact assessment for a landfill site. Furthermore the rate at which an emission occurs (its *flux*) is not considered in other processes (production of materials and energy). Only the overall magnitude of the emissions are known (as a pulse)\*. It would be illogical to consider the flux of a landfill site or the emissions from other processes as a pulse.

However, the quality of the landfill site will affect the flux of a substance. The situation becomes more complicated when degradable substances are considered. In this event two processes will occur simultaneously: the degradation of the substance in the landfill and emission into the environment due to leaks from the landfill site. The quantity of the substance which is actually introduced into the environment can be estimated by creating and solving a set of differential equations. This quantity can then be considered as the emissions associated with the landfill process.

The mass of a given substance present in a landfill site at time  $t$  is indicated by  $s(t)$ . The following applies to the mass  $da/dt$  which is degraded per time unit:

$$\frac{da}{dt} = \alpha s(t) \quad (2.8)$$

where  $\alpha$  is a constant which is inversely proportional to the half-life of the substance†. The following applies to the mass  $dm/dt$  released per time unit from the landfill into the environment:

$$\frac{dm}{dt} = \mu s(t) \quad (2.9)$$

where  $\mu$  is proportional to the permeability of the landfill site and will depend on the substance. It is assumed that the total quantity present  $s(t)$  is only removed by degradation or emissions:

$$\frac{ds}{dt} = -\frac{da}{dt} - \frac{dm}{dt} \quad (2.10)$$

thus

$$\frac{ds}{dt} = -\alpha s(t) - \mu s(t) \quad (2.11)$$

The solution to this equation is:

$$s(t) = s(0)e^{-(\alpha+\mu)t} \quad (2.12)$$

where  $s(0)$  represents the quantity of the substance present at  $t = 0$ , i.e. the quantity landfilled. Thus for the emissions to the environment during a period  $dt$

$$dm = \mu s(0)e^{-(\alpha+\mu)t} dt \quad (2.13)$$

The total emissions to the environment are determined by integration from  $t = 0$  to  $t = \infty$ :

$$m(\infty) = \int_0^{\infty} dm = \frac{\mu}{\alpha + \mu} s(0) = f_m s(0) \quad (2.14)$$

The emissions to the environment of a substance in a landfill site where it is degraded are thus found by multiplying the quantity landfilled by the factor  $\mu/(\alpha+\mu)$ . This factor (which could be called the *emission factor*‡ is indicated by  $f_m$  and has the following properties:

- $f_m$  is dimensionless;
- $0 \leq f_m \leq 1$ ;
- $f_m = 0$  if  $\mu = 0$ , i.e. when the landfill site is absolutely leakproof or if the waste is permanently encapsulated;
- $f_m = 1$  if  $\alpha = 0$ , i.e. if the substance is persistent.

The emissions into the environment could be divided into atmospheric emissions and emissions to

\* One of the consequences of this principle is the introduction of the HTP (Guinée & Heijungs, 1992).

† Actually:  $\alpha DT_{50} = \ln 2$ , where  $DT_{50}$  is the half-life of the substance under landfill conditions.

‡ The term emission factor is also used in another context, as a process parameter of environmental interventions (see *Handbook of emission factors*).



the soil. This would lead to a total of three relevant parameters: the degradation constant  $\alpha$ , the permeability to the soil (and groundwater)  $\beta$  and the permeability to air  $\lambda$ . In this event two emission factors will be required, one for air ( $f_i = \lambda/(\alpha+\lambda+\beta)$ ) and one for soil ( $f_b = \beta/(\alpha+\lambda+\beta)$ ).

In many cases the substance property associated with its degradability will be known from laboratory investigations. Under landfill conditions however, a substance may behave completely differently. The leak rate will normally also be unknown. It may be possible to estimate this on the basis of evaporation and leaching rates. A provisional solution for such cases might be provided by reduction to a few extreme situations:

- $f_m = 0$  for degradable or immobile substances in landfill sites which meet IBC criteria;
- $f_m = 1$  for landfill sites which do not meet IBC criteria;
- $f_m = 1$  for persistent substances.

Further research into the emission factors for landfilling is therefore necessary.

### 2.2.8 Qualitative environmental interventions

The list of environmental interventions can become very long. Subsonic vibration, fragmentation (road construction), defragmentation (land consolidation), weak electromagnetic fields (in the vicinity of electrical equipment and transmission lines), intentional damage (chemical and mechanical weed control) are all interesting matters which may have serious consequences. However, as they are not generally recognized environmental problems and are difficult to apportion to the functioning of a product they will not be collected as process data in practical studies. Where necessary a solution can be provided by giving a qualitative indication\* of certain interventions.

Here qualitative environmental interventions are environmental interventions which cannot be quantified sufficiently, if at all. As the aim is to conclude the inventory analysis with a table of measurable interventions the intention is to quantify as much as possible: physical interventions can always be measured in kg and other SI units. Where unquantifiable aspects are involved it is often unclear whether they are an intervention or an effect. These difficult cases will be discussed in more detail in the section on quantitative environmental aspects (§3.2.19).

## 2.3 The format

An LCA requires data about the processes included in the process tree. The *format* for the storage and presentation of this data is developed in this section. The format is a framework to indicate the structure of the process data and the information to be stored.

The overall structure, the *conceptual format*, will be discussed here. Besides the conceptual format there is also a detailed specification, the *(computer) technical format*. The technical format deals with aspects such as the use of decimal points and commas. This level of detail will not be covered here. The agreements which affect the way the data is entered will be discussed here.

At present it is not possible to set up a definitive conceptual format. There are two aspects which are closely connected. These aspects could be viewed as a form to be filled in: the first aspect is the actual form, the second the information entered in it. The more information is pre-printed the greater the "uniformity" will be, but not all responses can be pre-printed†. An example of the hierarchy of the process data to be entered in the format follows below (see also Table 2.2):

```

output
output to the environment
  emission to the air
    emission of aliphatic hydrocarbons
    emission of olefins
  
```

\* Qualitative statements are often called "flags".

† Example: a form may include the question "personal situation:". The information to be provided becomes much clearer if the question is changed to: "personal situation: (1) marital status, (2) number of children:". The question "marital status:" could be differentiated to "married", "single", etc.



emission of propene

Emissions to the air are the most detailed level in the form included in the appendix of the guide. The form would be greatly enlarged by the inclusion of even one more level. This section will now focus on the way in which the levels not specified in the form have to be entered.

The development of the LCA method requires making detailed agreements about the data storage, i.e. the format, to ensure uniformity and transparency. The format makes it possible to establish whether the difference in outcome of similar LCAs is due to a difference in the method or a difference in the underlying data. In essence the method imposes a set of constraints on the format.

The method is divided into a number of components: goal definition, inventory analysis, classification, evaluation and improvement analysis. Each component contains a number of sequential steps. The format for the storage of process data is important when the process data are entered in the second step of the inventory analysis. However, the format also has links with other steps in the method. This will have to be borne in mind when developing the format. The requirements which the format has to fulfil will be discussed in this section.

### 2.3.1 General format requirements

The format specifies the process data to be included in the inventory analysis and the way in which the process data are structured. This requires a more detailed consideration of the processes and process data in general.

An LCA requires a range of data such as the "relevant" quantitative data on environmental interventions and economic inputs and outputs. Qualitative data should also be included such as a description of the process and comments on the status, support, accuracy and process data allocation.

The term "process" is not strictly defined in the LCA. At present the guide does not provide a definition. In practice the level of detail will have to be limited. Without this limitation each individual machine in a factory would result in a separate process. In practice the term "process" can be applied effectively to industrial production plants (which are also covered by that term in the emission registration project; see also the *emission registration handbook*).

However, there is no definition of the breadth of the term "process" which is comparable to the definition of the depth. In addition to industrial processes there are many other processes: transport, usage (e.g. the use of a TV) and waste processing (e.g. the incineration of waste products with domestic refuse). Economic processes span our complete society.

Thus, in principle an LCA should include a wide range of data about potentially relevant environmental interventions in the analysis, and particularly economic inputs and outputs. The format should be suitable to record such a wide range.

To correspond with the method the format should also be suitable for aggregated processes which represent a number of interconnected processes. This requirement should not pose a problem given to modular nature of the process tree.

Given this consideration of the processes and process data the method imposes a set of constraints on the format.

- The format should be compatible with the process data, particularly those which are important in the classification. The further subdivision of environmental interventions and the required detailing of the relevant data such as substances and groups of substances forms an important element in this.
- The format should be compatible with the potentially relevant inputs and outputs: all inputs and outputs of a process which refer to other processes in the product chain. The detailing and subdivision is also important in this data. In this respect the format also depends on the system delineation of the product system.

We have seen that both the classification and the system delineation in the method affect the format. Thus the format not only concerns the step "process data inventory analysis" but it is also linked to other steps and stages in the method\*.

\* In principle this demonstrates that there is considerable feedback between the process data inventory analysis and other steps and stages. Such feedback also occurs in other steps.

There are some other requirements, besides the methodological delineation of the format.

- The format should relate to the actual availability of data. It would not be useful to apply an extensive classification if certain aspects can never be completed due to lack of data.
- Where possible the format should be compatible with normal practice.
- The format should be understandable and internationally applicable. It should be unambiguous and not lead to questions during the inventory analysis.
- The format should prevent an LCA from being overwhelmed by an excessive amount of process data to be entered.

The common requirements and practical constraints imposed on the format will result in an initial rough definition of the format. The requirements imposed on the format will be developed in this chapter.

### 2.3.2 Requirements related to environmental interventions

The method indicates the environmental effects to be considered during the classification. A list of all main categories of environmental interventions relevant to the format which the inventory analysis should cover can be drawn up on the basis of the description of the classification (see Table 2.1).

TABLE 2.1. Overview of environmental effects and associated environmental interventions.

environmental effect	environmental intervention	unit
depletion of abiotic resources	use of resources	kg, m <sup>3</sup>
depletion of biotic resources	use of resources	kg, ha, —
enhanced greenhouse effect	emissions to the air	kg
depletion of the ozone layer	emissions to the air	kg
human toxicity	emissions to the air	kg
	emissions to water	kg
	emissions to the soil	kg
ecotoxicity	emissions to water	kg
	emissions to the soil	kg
photochemical oxidant formation	emissions to the air	kg
acidification	emissions to the air	kg
nutrification	emissions to the air	kg
	emissions to water	kg
	emissions to the soil	kg
waste heat	heat emissions to water	MJ
noise	noise production	Pa <sup>2</sup> ·s
damage to ecosystems and landscapes	space use	m <sup>2</sup> ·s
victims	victims	—

A final classification of the relevant environmental interventions is not yet possible. The reason for this is that the relevant environmental interventions leading to certain environmental effects, e.g. radiation, are not yet clear. The method requires further development in this area. Naturally it is possible to compile a provisional list of groups of relevant environmental interventions.

When studying environmental interventions inputs and outputs into the environment have to be distinguished, similar to the distinction between economic inputs and outputs. Where environmental interventions such as space use are considered this distinction is not always clear. Both the use of resources and the use of space are environmental inputs, according to the terminology of the guide. The other environmental interventions in Table 2.1 are environmental outputs.



The table shows that some environmental interventions or groups of interventions contribute to more than one environmental effect. As there is no classification method for certain environmental interventions such as radiation it is not possible to identify the substances or groups of substances for which these environmental effects have to be included in the inventory analysis.

The classification of interventions into categories should not be unnecessarily detailed: the format shows that there are emissions to the air but not that there are acidifying emissions to the air. This will be shown by the classification. The distinction between biotic and abiotic resources is irrelevant for the same reason. However this will be included as a *aide-mémoire* (listing both biotic and abiotic resources), although the exact location of a resource in the format is irrelevant. Such classifications are no more than a guide during the inventory analysis and are not binding.

In principle each group of environmental interventions, particularly emissions, includes substances and groups of substances. Emissions into the atmosphere, for example, include emissions of CO<sub>2</sub>, CO and NO<sub>x</sub>. Each substance has a different impact on each environmental effect. CO<sub>2</sub> emissions into the atmosphere, for example, are important to the greenhouse effect but do not affect odours. Some substances are relevant to several environmental effects. NO<sub>x</sub> emissions into the atmosphere are relevant when assessing human toxicity, acidification and eutrophication. In principle the distinction between substances by their effects in the classification should not affect the format as this aspect is only relevant to the classification.

All that is relevant to the format is whether a given substance or group of substances has to be included in the inventory analysis. The guide includes a list of classification factors of relevant substances for some environmental effects. This can be used to determine whether a substance has to be included. However, these lists are not yet complete. The guide only lists a small number of relevant substances. For example, the ADI has been determined for more substances than are listed in the guide and the HCA could also be determined for these substances. Furthermore the lists do not include substances for which classification factors have not yet been determined but which, given their effects, should be included in the classification.

Eventually a complete list of emitted substances will include thousands of substances. Even then the classification of many substances would still be questionable due to the lack of reliable classification factors.

Hence in principle (but not in practice) the inventory analysis of emissions should cover more substances than are included in the lists in the guide. The relevant substances emitted by a given process will depend on the process concerned. Often only a limited number of parameters will be recorded for each individual process. In essence the emissions of all other substances are assumed to be zero\*.

Thus, the complete list of potentially relevant emitted substances is not yet available and can only be drawn up by carrying out a great many LCAs. However, this falls outside the scope of this study. It is likely that even then only an evolving list of substances can be drawn up which will grow as more LCAs are carried out. The initial list included in the appendix of classification factors to the guide can be used as a start. Two constraints are imposed on the list of substances:

- uniform substance names: many chemical substances have synonyms. A clear nomenclature will have to be agreed upon;
- avoid double inclusion in the classification: in the inventory analysis substance data may be included by substance or by group of substances. The classification is based on the assessment of data on groups of substances and individual substances. The classification of data during the inventory analysis should not lead to data being included twice in the classification.

For the time being these considerations will suffice. They can be used as guidelines in the compilation of a list of substances for the inventory analysis as well as the classification.

### 2.3.3 Requirements related to economic inputs and outputs

In principle every process in a product system is included in the process tree. Each process has certain

\* Negative emissions can occur in exceptional cases. A production forest results in negative CO<sub>2</sub> emissions to the air. Some crops lead to negative phosphate emissions.



economic inputs and outputs. There is a wide range of economic flows: these cover all forms of goods, materials, energy, services such as transport and the waste to be processed (waste which does not yet have a final destination and which will only be processed in a following process).

By definition the economic output of one process provides the economic input of one or more other processes\*. In this respect the economic flows are essentially the links connecting the processes in the process tree. Thus, economic flows are, in principle, symmetrical: all economic inputs may be the economic outputs of other processes.

However, given the system delineation method used in the guide a conflict arises between the clear definition of economic flows (during the inventory analysis) and a practical way to delineate the product system. The problem regarding the format is that, initially, it is not possible to determine how detailed the inventory analysis of the economic flows in the processes should be. In this respect the system delineation provides ample opportunity for interpretation by those implementing the LCA. A related problem is that the level of detail of an economic process may vary considerably for a given process, even if the overall picture of the economic flows of the process is known.

Let us assume that the economic flows of a process have already been surveyed in considerable detail. The reason for providing such detail might be that the process and its economic output are a relatively important element in the process tree. After such a detailed inventory analysis the economic flows of the process should be known, even if the process is included in other process trees for other LCAs.

In principle the problem referred to above could be solved by including the complete and detailed data of that process (where available) as well as all other process data in a database accessible to all. These complete process data could, and should, then be included automatically in any later LCA pertaining to the process. Aggregated processes could also be used.

For the time being this solution is not feasible. Hence, the level of detail of the economic flows will continue to be directly linked to the delineation of the product system. For a given process this delineation may be different in each LCA. As a result of the system delineation some flows will not lead to subsequent processes in the process tree. Instead they will be cut off (noted as *p.m.*). The problem is that this occurs at the product system level rather than at the process level.

Thus the format for economic data should be based on all flows. The format should allow for the most detailed inventory possible of all the economic flows of a process. After entering this data some simplifications will be made, depending on the product system under consideration. Certain items will be noted as *p.m.* to prune the process tree to manageable proportions.

The guide provides a rough classification of the economic flows: the division into goods, materials, services, energy and waste to be processed. Strictly speaking, this rough classification is not clear as there is no proper distinction between goods, materials and waste to be processed, etc. Hence even this rough classification can only be used as an aid when carrying out LCAs and is not binding. There is no overview of all possible economic data for all processes. This list would be very large: it is to be expected that hundreds of thousands of items could be distinguished. The lack of such an overview need not impede LCAs. However, such an overview would make the method much more transparent. Note that, strictly speaking, according to the method, each economic output is the result of at least one process. An overview of economic flows is tied directly to an overview of the processes.

An overview of possible economic flows as well as a list of substances for environmental interventions could be developed by carrying out many LCAs. The following aspects are pertinent to the overview of economic flows:

- each overview is incomplete and it is unlikely that there will ever to be a complete overview;
- the nomenclature for economic data should be standardized;

The compilation of such a list, however, is beyond the scope of this project.

\* The only exception to this rule is formed by the economic input representing the functional unit. However, in principle, this output can be used as an input in another process.

### 2.3.4 Other constraints

There are other constraints on the format than those arising directly from the delineation of the product system and classification. The data format should be clear with respect to the completeness of the mass and energy balances of the process. The format should also ensure that the process data is entered so that nothing is included twice, either when completing the balances or later in the classification. The format for storing process data should also offer facilities for a further description of the quality and nature of the process data. The effect of these aspects on the format will be discussed below.

#### The mass balance

Given the requirement of consistency the mass balance\* for each process should be complete:

$$\sum_i m_{in,i} = \sum_j m_{out,j} \quad (2.15)$$

where  $m_{in,i}$  is the mass of the  $i^{\text{th}}$  input and  $m_{out,j}$  the mass of the  $j^{\text{th}}$  output†. In practice the mass balance will not be complete when the first attempt is made to enter the process data. This will probably be due to one of the following causes:

- the process data are incomplete;
- something has been included twice;
- the various data have been rounded or the orders of magnitude vary too much‡.

In practice the lack of data will lead to significant hiatuses in the mass balance, for example when the required quantity of fuel for a combustion process is specified as well as the quantity of residual material and the total quantity of flue gasses but not the necessary amount of air for the combustion (i.e. the quantity of oxygen required for combustion + the residual air + any excess air).

Double inclusion can creep into data about environmental interventions with relative ease. For example, an item "ethanol emissions" may also have been included under the general heading (subtotal) "hydrocarbon emissions". If this is overlooked it will be included twice when a mass balance is drawn up. How to avoid double inclusion is discussed below.

If the various process data differ widely (in order of magnitude or accuracy, or because the data comes from different sources) the hiatuses in the mass balance will generally be relatively large.

The mass balance is a verification aid in LCAs (double inclusion and missing data) and is also used to augment the data§. In principle the mass balance for each process will be complete if all data is available, as mass cannot be lost. An *estimate* will have to be included if the mass balance in the inventory analysis is still not complete even after augmentation. This item is included in the format outside the environmental interventions and the economic flows. This estimate indicates the incompleteness and/or inaccuracy of the process data and is important during the evaluation (step 4.2).

The mass balance is used for environmental interventions as well as economic flows. Not all process data are relevant to the mass balance, examples include electricity consumption and chemical oxygen demand (COD). Some process data will have to be converted to mass data (m<sup>3</sup> flue gas to kg flue gas, number of caps to kg caps, etc.).

It is advisable to specify all items in the mass balance separately and to indicate any conversion factors used.

#### The energy balance

Given the consistency requirement the energy balance for every process will have to be complete. This is also important for the classification as the energy related environmental aspects (such as heat losses and fossil fuel depletion) are also classified.

\* It is often advisable to draw up some mass balances of elements, e.g. carbon or various heavy metals, in addition to the general mass balance.

† The requirement that mass balances are complete in LCAs is internationally accepted (SETAC, 1992).

‡ Economic flows are often expressed in tonnes while the environmental interventions are on a kg or mg scale.

§ For some chemical and biological processes it may be useful to draw up one or more atomic balances (e.g. a carbon balance) in addition to the mass balance. Again in principle the balance should be complete.



$$\sum_i E_{in,i} = \sum_j E_{out,j} \quad (2.16)$$

where  $E_{in,i}$  is the energy of the  $i^{\text{th}}$  input and  $E_{out,j}$  the energy of the  $j^{\text{th}}$  output. In practice the energy balance will often be incomplete. There is a number of possible causes of this.

- Many energy related aspects such as evaporation, chemical conversion and combustion have to be considered. In practice it is often impossible to draw up the complete energy balance given the lack of data about the energy content of and the chemical energy stored in the various inputs and outputs.
- There is no uniform definition of the term energy content. In practice different concepts are often used for the energy content of materials. The Gibbs energy (the free energy of materials, substances and mixtures) is the correct approach in thermodynamic terms. However, other quantities are often used such as the heat of combustion or the gross energy requirement (GER). For the GER the energy content is traced back to the heat of combustion of fossil fuels using conversion efficiencies and by taking into account the underlying processes and assumed conversion efficiencies. With the available energy indicators it is not always clear what they refer to.
- In principle the energy content is a function of temperature and pressure. Hence a reference level (temperature and pressure) will have to be defined. However, in general discrepancies will only lead to minor hiatuses in the energy balance.

Thus, the energy balance of a process is a thermodynamic calculation and requires extensive data about energy, entropy, etc. Often this data is unavailable.

Given the problems associated with the creation of the complete energy balance we will often have to limit ourselves to a *global/overall/approximate energy balance* of a process. The following items may be included in the global energy balance:

- energy input and output via electricity and power;
- energy input and output through steam, hot gasses, etc.;
- energy and heat created by combustion processes;
- heat lost through water and air and possibly by heat radiation;
- heat released or energy taken up by chemical processes, melting, evaporation, etc.

In principle the first three items in a global energy balance are derived directly from the surveyed economic flows and environmental interventions (through the heat of combustion, etc.). The fourth item is derived from the emissions (into the atmosphere and water) surveyed based on the calculation of the first three items in view of the conversion efficiency and heat losses in the process. The last item refers to energy related aspects for which sufficient data will rarely be available. In practice this item can only be estimated to complete the global energy balance. Therefore the format for the process data will include one estimate.

The global energy balance is primarily an instrument to check and supplement the available process data. It is also used to make a global estimate of the heat loss data and to quantify the take up and release of energy by materials and substances. Energy may occur in the form of environmental interventions as well as economic flows. Not all process data affect the global energy balance. The global energy balance will also require additional data. Some process data have to be converted using the heat of combustion of substances or the energy content of energy carriers (steam, hot gasses, etc.). The global energy balance also requires process-specific data about conversion efficiencies, heat losses, etc. Like the mass balance the global energy balance itself is not included in the format for the storage of process data\*. The global energy balance is a calculation based on a selected set of process data from the inventory analysis. It is therefore advisable to indicate all items in each process which are included in the energy balance, as well as any conversion factors.

In practice not all the items in the global energy balance will have to be worked out. For example, the heat of combustion of the process input does not need to be determined if combustion does not

\* This is still under discussion internationally. During the SETAC workshop in Leiden in 1991 it was indicated that parallel to this discussion, the energy balance requires its own format. The current format includes, for example, an input of  $x$  kg steam, while the energy format would list an input of  $y$  kJ steam.



occur during the process. This will be the same as the total of the outputs.

Thus, the flows which contribute to the global energy balance will be different in practice for each process.

#### *Avoiding double inclusion of data*

Various forms of double inclusion of data may occur during the inventory analysis and the subsequent classification of the process data:

- double inclusion of process data during the classification;
- double inclusion of process data in the mass balance.

Data about environmental interventions such as emissions are generally specified by substance. However, sometimes data is provided by group of substances (mixtures). For example, ethanol emissions may be specified at the substance level and the emission of all alcohols (i.e. including ethanol) is also specified. In this case the latter item is a subtotal for the emission of alcohols. The classification may be based on either the substance group or on the individual substances. For example, different POCPs may have been defined for the group alcohols and ethanol individually. If both items were classified ethanol would be included twice\*.

In principle this problem could be avoided by following the method described below. However this requires a complete list of substances and a classification of the emissions. Let us assume that the classification used in Table 2.2 is used. The method is as follows: if data and certain substances have been included in the inventory analysis at the substance levels and are then classified classification based on a higher level category or sub-category is generally not permitted to avoid double inclusion of data.

In the example of ethanol it is possible to take the emission of the various alcohols (methanol, propanol, and others besides ethanol†) into account, but the sub-category alcohols and the higher level category substituted aliphatic hydrocarbons, should not be included in the classification. As a result the item "unspecified alcohols" is very important.

Other sub-categories in the example could be classified, e.g. chlorinated aliphatic hydrocarbons, unless individual substances (e.g. methylene chloride) are specified separately. However as yet this method and the classification used in Table 2.2 cannot be fully implemented. The reason for this was referred to in §2.3.2. At present there is no complete list of all categories and substances for individual environmental interventions.

The inventory analysis of data about substances as well as groups of substances can easily lead to double inclusion of data in the mass balance. The method described above as well as a systematic approach in the classification will reduce the risk of double inclusion in the mass balance as well.

Special care has to be exercised when classifying emissions as shown in Table 2.2, particularly when dealing with specific emissions into the atmosphere, water and soil. For example, an item like COD should not be included in the mass balance, or only partly. An item such as dust emissions might also include part of the heavy metal emissions. Generally the economic inputs will affect the mass balance. A uniform classification of economic flows, which is not yet available, would greatly reduce the risk of double inclusion.

With the exception of electricity and power all items relevant to the energy balance are derived from mass items (economic input or environmental intervention, e.g. steam, gas, fuel oil, liquid nitrogen, etc.). To avoid double inclusion of data in the process tree the global energy balance should not be linked to earlier or later processes in the tree when it is drawn up. These links are established for associated mass items. Only the separate energy items included as such in the economic flows (electricity in kWh and power in MJ) are linked to other processes in the production system.

\* In this context double inclusion of data during the classification does not refer to the classification of a substance under more than one environmental effect. Alcohol emissions for example, may have several environmental effects such as photochemical smog formation and human toxicity (see Chapter 3).

† This item includes the emission of unspecified alcohols, hence it has to be classified.

TABLE 2.2. Categories of emissions.

environmental intervention	category	sub-category	substance category	
total emissions	aliphatic hydrocarbons	methane		
		unspecified saturated aliphatic hydrocarbons	ethane	
			propane	
			...	
			unspecified saturated aliphatic hydrocarbons	
		olefins	ethene	
			propene	
			...	
			unspecified olefins	
			unspecified aliphatic hydrocarbons	
		substituted aliphatic hydrocarbons	alcohols	methanol
				ethanol
				propanol
				...
				unspecified alcohols
	aldehydes	formaldehyde		
		acetaldehyde		
		...		
		unspecified aldehydes		
		chlorinated alkanes		
...	...	...	...	

#### Data for the allocation rules

The process data is allocated after it has been entered (see §2.1.3). The subflows (flows with a negative value) of multiple processes are divided among the main flows (flows with a positive value).

If there is co-production all subflows are divided among the main flows, these will include a number of economic outflows. The flows allocated to must be specified, in other words, the main flows have to be identified. In §2.3.3 (analysis of the economic flows) it was mentioned that the classification into categories such as materials and goods is not uniform. In principle the economic outputs of a process which are not main flows could also be classified as "waste to be processed", for example.

In the section about allocation it was mentioned that economic data is sometimes required for the allocation. It was also indicated that the allocation cannot be based on the same principles for all processes. Thus, separate allocation rules have to be used for some processes. This has to be defined in the format, together with the relevant process data. The allocation itself is not included in the data stored, however the format may include the necessary markers and information relevant to the allocation. Sometimes the allocation method is specified for each economic output. These special aspects of allocation have not yet been developed in sufficient detail. Hence it is not yet clear what data is required. For the time being we will just have to bear in mind that the format for these processes must be sufficiently flexible to provide for the economic flows. This may involve more than

just flagging some items.

#### *Description of the nature and quality of the process data*

Some process data will be collected specifically for the products to be investigated and some will be generally available in various files. The same type of quality requirements and representativeness criteria apply to both types of process data. When general basic files are set up there is no process tree in which methodological choices have to be made or justified. When creating the basic files methodological choices should be avoided wherever possible. The cumulative aggregation of process data to a higher level process provides a test of the value of the selected process delineations. An example is provided by the process "the production of rolled steel" as the umbrella process covering all processes required for the production of a given quantity of rolled steel: from the extraction of resources via steel making, alloying, rolling and surface treatment to the completed product. Such an aggregation implies setting up a sub-process tree. This sub-process tree should also meet the general requirements for process trees. The methodological choices made should be specified and, ideally, justified.

The requirements are defined in the form of a specification of the

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

The scale level, global data, an indication of duration or capacity and the status of the process are all relevant to the *representativeness*.

A process or coherent group of processes may be representative for different spatial *scale levels*, from the global, western, national and local levels to the level of an individual company. The status needs to be specified so that the proper process can be selected in a given life cycle assessment. The reason for this is that a study of this kind can only be carried out for a specified level of scale.

The *approximate date* does not concern the exact data of the primary sources (see below) but rather the period for which the process is representative. A prospective study is about future processes, a product comparison of existing products is about current processes and a historical comparison is about historical processes.

The *duration of a process* refers to the time required to process the specified quantity. It is an interesting parameter in relation to the *capacity* of the plant. The inventory analysis of some process data (space use and noise) also requires this period to link the quantity of space used and noise generated by a process to a product.

Process data may refer to processes for which actual measurements have been made. However, a theoretical and hypothetical process such as a conceptual design for a future process or a definition of the current best practicable means may be used. Only allocated data may be available for a process or the data may have been obtained by extrapolation or from models. This determines the *status* of the process.

The *quality* of the process data can be described on the basis of the clarity of the process definition, the correctness and completeness of the data and the nature of the sources.

When examining the *clarity of the definition* of the processes a distinction has to be made between the internal definition, in terms of inputs and outputs, and the external definition, which specifies relationships with other processes. The latter is largely concerned with the delineation and inclusiveness of the relevant processes. This is illustrated by the following example. Coke production also produces ammonia in aqueous solution which is discharged into the sewers. The extent to which this process output becomes an emission depends on any sewage treatment plant which may be present. If the specification is limited to "x kg ammonia in water" the process description will be ambiguous. A process description should indicate whether it refers to the normal operation or whether emergencies, incidents, maintenance, effects of depreciation of capital goods, administration, staff facilities, etc. have been included in the quantification of the inputs and outputs. The choices made here will determine what may need to be added to a defined process. To make a balanced comparison possible the choices should all be made in a similar manner.

In general the *accuracy of the data* cannot be directly verified. The only option is to compare the



data with other independent sources. It is likely that the data is accurate if comparison reveals only minor discrepancies. The mass and energy balances can also be used as verification tools (see the discussion about mass and energy balances above).

The process specifications should be *complete*, all relevant economic inputs and outputs: products and by-products, including capital goods, energy and auxiliary materials (with positive values) and waste (with a negative economic value). The expected processing method of waste to be processed, particularly within a company, should also be specified. All environmental interventions used in the appraisal should be specified, as well as the economic inputs and outputs. For many processes some data will be known, some will be unknown and many inputs and outputs will be zero. Data which is not listed is the most unclear: is it zero or not available? Checking for completeness requires a practical knowledge of the field concerned. Again, material and energy balances can be used to check the main flows in a process.

The age of the data cannot be derived from the date of publication of the *source*. The research on which primary publications is based is often several years old, in secondary publications it may be as much as several decades. Data from different years is often used in a process description which may lead to inconsistency. For this reason the age of each data item or group of data should be specified. The authority of the data depends on the nature of the source. Primary and secondary sources have to be distinguished. Primary sources may be classified in three categories, of decreasing authority: independent, dependent and externally appraised, dependent but not appraised. Secondary sources can improve the status of the data if several primary sources are compared and their consistency appraised. Naturally, an *overall appraisal* can be made on the basis of the above criteria.

The format to store the process data should provide for a more detailed description of the quality, source and nature of the process data. The evaluation of an LCA includes a reliability analysis (step 4.2) for which this information is required. Such information is also important for the management of the final file with process data.

The above criteria provide information about the *nature and quality of the process data on average*. The accuracy of the inventoried process data *within a process* will vary widely in terms of source, support, etc. These aspects will have to be specified for each data item if there are significant deviations from the average criteria. This is relevant to making a correct final interpretation of the process data in the evaluation stage of an LCA. Thus, in some cases an average indication will be insufficient.

It is not yet clear to what extent this will require modification of the format. For the time being we will have to note that the format should provide for this. In addition to being able to give an indication of the accuracy, support, etc. this may also require a facility to specify the range of individual data. This would make it possible to perform a reliability analysis in which the range of the data produces a range in the results (step 4.2).

### 2.3.5 Recommendations for further development of the format

Some constraints on the format for storing the process data were analysed above. Provided all the requirements and notes are considered the provisional format specified in the guide may be used.

Naturally the process data may be specified in different units. It is advisable to use the *Système International des Unités* (SI) for all units. Economic flows and environmental interventions generally relate to quantities, i.e. data on mass, volume or number of items. Energy is specified in kWh (electricity) or MJ (e.g. heat). Space use is expressed in m<sup>2</sup>·s and noise production in Pa<sup>2</sup>·s as discussed in the section on process data (§2.2).

At present there are no units for the descriptive, qualitative data hence there is still scope for individual interpretation.

The following notes apply to the format for the inventory analysis of process data as given in the guide:

- Lists of substances: the importance of uniform lists of substances for the various environmental interventions, particularly emissions, was indicated earlier. The lists of classification factors given in the guide are still incomplete and need to be developed through use in practice, along with the classification. The lists of substances can be developed further on the basis of the lists used in the Emission Registration project. Its link with the list of classification factors imposes

a constraint on this.

- Lists of products and materials: the format of the economic inputs corresponds with that of the economic outputs. A list of all economic flows which in principle can be distinguished is not available. However an overview, even if incomplete, would be valuable.
- Process lists: there is no list of all the basic and aggregated processes which can be distinguished. This list would be directly linked to the overview of economic flows referred to above. Hence a step towards a list of products would also be a step towards a list of processes.

## 2.4 The inventory table

Three elements are discussed in the section on the creation of the inventory table: the calculation of the inventory table, the levels of aggregation and the presentation of the inventory table.

### 2.4.1 Calculation of the inventory table

After the collection, and where necessary allocation, of the process data the extent to which each of the processes occur in the process tree has to be determined. We will discuss two methods for this:

- the sequential method;
- the matrix method.

In the *sequential method* the calculations are made on the basis of the functional unit. References are made to earlier processes in a certain ratio, based on the product, material and energy needs of the process which supplies the functional unit. The defined waste processing services are also provided in the right quantity, based on the waste produced by the functional unit. In turn the requirements of these processes initiate the next round of calculations. In this way the entire chain is followed until there is no further demand or supply of product, materials, etc. This is of course, why this approach is known as the sequential method.

The sequential method is most commonly used. This method is used by the SIMAPRO program developed at CML (Goedkoop & Jansen, 1991) and the commercial program IDEA (Lübker *et al.*, 1991). This method is also referred to in a report on a workshop about inventory analysis in LCAs, published by the American branch of the *Society of Environmental Toxicology and Chemistry* (SETAC) (Fava *et al.*, 1991). One problem with this method is that there is no quick and accurate way to deal with processes which refer to themselves or which have mutual references (recursion\*). In SIMAPRO this causes a stack overflow† whereas SETAC recommends a limited number of iterations and IDEA more accurately develops convergence criteria as a stop condition.

The most basic example of recursion in a process tree will be discussed below. Let us assume that a process requires 5 MJ of electricity and that a process "electricity production" has been defined whose output includes 1 MJ of electricity and whose input includes 0.1 MJ electricity. In the sequential method the process is used after multiplying by 5 to supply the 5 MJ required. However, this requires 0.5 MJ and the process will be called another 0.5 times. This will then result in a demand for 0.05 MJ, etc., *ad infinitum*. As a result the total quantity of electricity required is  $5 + 0.5 + 0.05 + \dots = 5.5555\dots$ . The following theorem can be used:

$$y \sum_{i=0}^{\infty} x^i = \frac{y}{1-x} \quad (2.17)$$

where  $y = 5$  and  $x = 0.1$  to obtain 5.5555... directly, or by including the effective output (0.9 MJ) in the process specification to find out that the process is called 5.5555... times. However, this is more difficult for two or more processes which call each other. The processes to produce electricity and coal

\* The SETAC report (Fava *et al.*, 1991) refers to process groups in which recursion occurs as nonlinear networks. In this project the term "nonlinear" is only used for processes in which there is no constant ratio between input and output (see §2.2.1).

† This is because process A calls process B, which is followed by a call from B to A. A stop criterion in the form of a sufficient number of decimal places is not defined, hence the computer will continue the calculations forever.

are examples of this (Boustead, 1990).

In the *matrix method* the occurrence of the processes is determined simultaneously rather than in sequence\*. This makes it possible to deal with feedback immediately, rather than by jumping between processes. The matrix method has another advantage: it is possible to provide algebraic expressions for the quantification and occurrence of each process and thus for the inventory table and the environmental profile. The matrix method has been described by the Fraunhofer-Institut (ILV, 1991), and by Heijungs (1992). We will present the second description as this provides more suggestions for further analysis, i.e. the reliability analysis (§4.2), the dominance analysis (§5.1) and the marginal analysis (§5.2).

In the matrix method the data contained in the process tree is formed into a matrix. A process is represented by a column vector where the upper part ( $a_1, \dots, a_r$ ) contains the economic inputs and outputs†, and the lower part the inputs from and outputs to the environment ( $b_1, \dots, b_s$ ):

$$\begin{pmatrix} a \\ b \end{pmatrix} = \begin{pmatrix} a_1 \\ \dots \\ a_j \\ \dots \\ a_r \\ b_1 \\ \dots \\ b_k \\ \dots \\ b_s \end{pmatrix} \quad (2.18)$$

We will use the following sign convention: inputs are negative, outputs are positive.

By numbering all processes with an index  $i$  and by placing them in a matrix such that the data on each input or output is presented in a row, we obtain:

$$\begin{pmatrix} A \\ B \end{pmatrix} = \begin{pmatrix} a_{11} & \dots & a_{1i} & \dots & a_{1q} \\ \dots & \dots & \dots & \dots & \dots \\ a_{j1} & \dots & a_{ji} & \dots & a_{jq} \\ \dots & \dots & \dots & \dots & \dots \\ a_{r1} & \dots & a_{ri} & \dots & a_{rq} \\ b_{11} & \dots & b_{1i} & \dots & b_{1q} \\ \dots & \dots & \dots & \dots & \dots \\ b_{k1} & \dots & b_{ki} & \dots & b_{kq} \\ \dots & \dots & \dots & \dots & \dots \\ b_{s1} & \dots & b_{si} & \dots & b_{sq} \end{pmatrix} \quad (2.19)$$

The number of processes in the process tree is  $q$ . Rows represent flows of one type (e.g. kg PVC or MJ electricity), columns represent processes (e.g. glass production). The first input or output of the first process occurs in the upper left-hand corner of the matrix, the first input or output of the last ( $q^{\text{th}}$ )

\* The theory of matrices, determinants and linear algebra is discussed in all elementary texts on linear algebra (e.g. Apostol, 1969).

† Another option would be to categorize the interventions on the basis of the problem associated with them. For example:

• This method could also be referred to as the "simultaneous method" to stress the difference with the sequential method.  
 † In accordance with most common conventions matrixes will be represented by bold Greek capitals ( $X, Z$ ), vectors by bold lower case letters ( $x, \xi$ ), and scalar variables by italics ( $x, \xi$ ). Unfortunately there is very little difference between italics and bold italics, hence the meaning has to be derived from the context.



process in the upper right-hand corner.

The entire process tree provides an external function (see Figure 0.3): the functional unit selected in the goal definition. In vector notation this is presented as part of a *kernel process*; this is the process, possibly a fictitious one, containing the entire product system; its only economic output is the functional unit:

$$\begin{pmatrix} \alpha \\ \beta \end{pmatrix} = \begin{pmatrix} \alpha_1 \\ \dots \\ \alpha_j \\ \dots \\ \alpha_r \\ \beta_1 \\ \dots \\ \beta_k \\ \dots \\ \beta_s \end{pmatrix} \quad (2.20)$$

where  $(\alpha_1, \dots, \alpha_r)$  represents the economic part of the kernel process, and  $(\beta_1, \dots, \beta_s)$  the inventory table. The functional unit corresponds with one of the  $\alpha_j$ ; the other components of  $\alpha$  equal 0. The inventory table  $\beta$  is unknown: the goal of the inventory analysis is to calculate the inventory table.

The premise used when calculating the occurrence of each process is that all products, materials, services, etc. used by a process are supplied by other processes and that the process waste to be processed is handled by other processes. Hence there will be a balance for each row; depending on the nature of the row this could be a mass balance, energy balance or another balance such as a service balance. When the quantified contribution of each process is referred to as  $p_i$  the balance of the  $j^{\text{th}}$  row is given by:

$$\sum_{i=1}^q a_{ji} p_i = \alpha_j \quad (2.21)$$

This applies to each row in the economic section:

$$\forall j = 1, \dots, r: \sum_{i=1}^q a_{ji} p_i = \alpha_j \quad (2.22)$$

This equation is known as the *balance equation*.

Single processes are used when calculating the inventory table; multiple processes were allocated at an earlier stage. The consequence for the balance equation is that  $q = r$ : matrix  $A$  is square. However, the distinction between  $q$  and  $r$  will continue to be used to make the notation easier to understand.

According to linear algebra a linear system of equations such as the balance equation has a solution\* for the coefficients  $(p_1, \dots, p_q)$ , given by Cramer's rule:

$$p_i = \frac{\det(A^i)}{\det(A)}, \quad i = 1, \dots, q \quad (2.23)$$

Here  $A^i$  is matrix  $A$  where the  $i^{\text{th}}$  column has been replaced by  $\alpha$ :

\* The conditions are that the matrix should be square ( $q = r$ ) and that matrix  $A$  is not singular ( $\det(A) \neq 0$ ). The first condition is fulfilled by the allocation, in practice the second condition will also be fulfilled, otherwise the process tree would contain unnecessary or contradictory processes.

$$A^i = \begin{pmatrix} a_{11} & \dots & a_{1i-1} & \alpha_1 & a_{1i+1} & \dots & a_{1q} \\ \dots & \dots & \dots & \dots & \dots & \dots & \dots \\ a_{j1} & \dots & a_{ji-1} & \alpha_j & a_{ji+1} & \dots & a_{jq} \\ \dots & \dots & \dots & \dots & \dots & \dots & \dots \\ a_{r1} & \dots & a_{ri-1} & \alpha_r & a_{ri+1} & \dots & a_{rq} \end{pmatrix} \quad (2.24)$$

and  $\det(A)$  is the *determinant*\* of matrix  $A$ .

The inventory table can now be obtained by multiplying the coefficients  $p_i$ , obtained from (2.23), by the process characteristics  $b_{ki}$  and by totalling them for each environmental intervention. The reason for this is that they were, in effect, "included in the multiplication":

$$\beta_k = \sum_{i=1}^q b_{ki} p_i, \quad k = 1, \dots, s \quad (2.25)$$

The application of Cramer's rule ensures that processes in the process tree which refer to each other are included without iteration or cut-off. This is due to the fact that the equations in (2.22) are solved simultaneously by the matrix method. While in the sequential method the contribution of each method is calculated individually, without regard to their interdependence.

#### 2.4.2 Presentation of the inventory table

The inventory table can be specified at different levels:

- at the process level;
- at the substance level;
- at the product level.

These three forms can correspond with the process policy, the substance policy or the product policy, respectively. Only the third option will be developed within the framework of this study. Here a simple reference to the other forms is sufficient.

The presentation of the environmental interventions of a process, or group of processes, by ratio of the share of the process concerned can provide information about the share of that process, or group of processes, in a given environmental problem, particularly when an environmental profile is drawn up on the basis of this partial inventory table. In a substances policy this might result in recommendations for the use or introduction of other processes. This will be covered under the dominance analysis (§5.1).

The inventory table of a substance or group of substances may be pertinent to a substance-oriented environmental policy. For example, it could indicate heavy metal emissions during the life cycle of a product†.

At the product level, however, the inventory table can be used to influence policy on products, to which the remainder of this study is addressed.

\* The theory of matrices, determinants and linear algebra is discussed in all elementary texts on linear algebra (e.g. Apostol, 1969).

† Another option would be to categorize the interventions on the basis of the problems associated with them. For example a categorization could be made for acidifying substances, greenhouse gasses, etc. This could be regarded as a pre-classification. The problem which arises is that the categories overlap: a substance such as CFC-11 has to be included fully (in kg) under greenhouse gasses as well as under gasses which deplete the ozone layer. As a result the mass balance of the environmental interventions becomes confusing.

2.5 Discussion

The entire process tree provides an external function (see Figure 9.3): the functional unit selected in the goal definition. In vector notation this is presented as follows:

*"What a funny watch!", she remarked.*  
*"It tells the day of the month, and doesn't tell what o'clock it is!"*  
*"Why should it," muttered the hatter. "Does your watch tell you what year it is?"*

LEWIS CAROLL, Alice's adventures in Wonderland.

So far the inventory analysis is the LCA component which has received most attention worldwide. Undoubtedly this is because the goal definition is largely concerned with social aspects and the classification, evaluation and improvement analysis were only developed on the basis of the inventory analysis. However, this does not mean that the theory behind inventory analysis will soon be complete. On the contrary, some new aspects are developed in this report and other aspects have not yet been taken up. One example is the inclusion of landfilling waste as a process in the process tree, final waste has always been considered as an environmental effect in most studies. Inclusion in the process tree is relatively new (Guinée *et al.*, 1990; Guinée, 1992; Finnveden, 1992), but how to do this is not yet clear, although the concept of emission factors is suggested in this report. Examples of other debatable aspects include the allocation when co-production and open-loop recycling are used. Another difficult problem, although not so much concerned with the principles, is the cut-off of the process tree. Studies on more streamlined methods (see §1.2) in particular will continue to influence both theory and practice for some time to come.

The formulation of clear mathematical formulas for LCA calculations means that there is a growing need, as well as a growing opportunity, for the use of software. However, for the time being the availability of process data will continue to be limited. Many of those involved in LCAs have clearly voiced the need for a central database, and expressed their willingness to help create such a database.

These three forms can correspond with the process policy, the substance policy or the product policy, respectively. Only the third option will be developed within the framework of this study. Here a simple reference to the other forms is sufficient.

The presentation of the environmental interventions of a process, or group of processes, by table of the share of the process concerned can provide information about the share of the process, or group of processes, in a given environmental problem, particularly when an environmental profile is drawn up on the basis of this partial inventory table. In a substance policy this might result in recommendations for the use or introduction of other processes. This will be covered under the dominance analysis (see §2.1).

The inventory table of a substance or group of substances may be partitioned in a substance-weighted environmental policy. For example, it could indicate heavy metal pollution during the life cycle of a product.

At the product level, however, the inventory table can be used to influence policy on products, which the remainder of this study is addressed.

$$A^{-1} = \frac{adj(A)}{det(A)} \quad (2.23)$$

Here  $A^{-1}$  is matrix  $A$  where the  $i^{\text{th}}$  column has been replaced by  $a_i$ .

The theory of matrices, determinants and linear algebra is discussed in all elementary texts on linear algebra (e.g. Apostol, 1989).

Another option would be to categorize the interventions on the basis of the problems associated with them. For example a categorization could be made for acidifying substances, greenhouse gases, etc. This could be regarded as a pre-classification. The problem which arises is that the categorization would be a subjective one and it is not clear how to do this. (e.g.) other procedures may be used to select substances which are most important. The most important substances in environmental interventions become categories.



## CHAPTER 3

# CLASSIFICATION

This chapter discusses how to proceed further with an LCA after the inventory analysis. Recently this aspect has received considerable attention (Ahbe *et al.*, 1990; Guinée, 1992; Hofstetter, 1991; Huppel & Guinée, 1992; Finnveden *et al.*, 1992). Some have asked whether it is actually necessary to undertake any steps beyond the inventory analysis (Braunschweig, 1992; Guinée & Udo de Haes, 1992). If all interventions in the inventory table pointed in one direction no further steps would be required in principle at least, to arrive at a conclusion. This was proposed as the conclusion of step 2.4. Naturally, the classification should not be started if there is no need for it.

However, in most cases some aspects of a product will be better while others will be worse. This applies to market comparisons as well as to comparisons of improved product alternatives. The prioritisation of effects is also desirable for the improvement analysis. Further information will then be required about the importance of the various environmental interventions. There is also a need to reduce the volume of data in the inventory table for assessment in certain applications such as environmental approvals, product information and eco-labelling. The additional information required includes empirical data about environmental processes and problems as well as a knowledge of socio-political weighting procedures. Two approaches can be taken on the basis of these two considerations.

In the first approach the two elements are combined in a single methodological component. This method is used in the Swiss *eco-points system* (Ahbe, 1991), the Swedish *EPS system* (Ryding, 1991), and the *DESC system* used in the Netherlands (Krozer, 1990). In the other approach the elements are dealt with separately in two separate methodological components, the *classification* and the *evaluation*. In the United States the classification and evaluation (US: valuation) are considered as steps in one component: the impact analysis. In this way various empirical and normative aspects can be separated as much as possible. One of the main reasons for this division is that the two types of aspects require different kinds of expertise.

In this way the environmental interventions could be aggregated in the *classification* using empirical information about environmental processes and problems wherever possible\*. The different types of environmental problems are compared during the evaluation using decision theory from the social sciences. The second approach in which the elements classification and evaluation are distinguished has been discussed in workshops on LCAs in Leiden, the Netherlands (Udo de Haes, 1992a; Udo de Haes, 1992b; Udo de Haes, 1992c) and Sandestin, US (Fava *et al.*, 1992).

As shown above it is not self-evident that these components should be separated. They are sometimes combined, in the Swiss method for example. In this method the interventions are all aggregated by calculating an *ecofactor* (eco-points/gr) for each type of intervention, multiplying this ecofactor by an intervention and then aggregating the result of all the interventions. This is expressed in the following formula:

\* In American terms: during the classification the measurement points (interventions) are aggregated on the basis of their potential contribution to a number of assessment points (problem types).

## 2.5 Discussion

$$ecofactor_i = \frac{1}{Fc_i} \times \frac{F_i}{Fc_i} \times c \quad (3.1)$$

where  $Fc_i$  represents the *critical flow* of substance  $i$ ,  $F_i$  the current volume of the intervention  $i$  in Switzerland and  $c$  is a dimensionless number used to increase the result. The critical flow  $Fc_i$  is based on emission standards for substance  $i$  which are based on a political decision. The number of eco-points is determined by multiplying the intervention by the ecofactor and the aggregation of the interventions:

$$total(\text{eco-points}) = \sum_i ecofactor_i \times m_i \quad (3.2)$$

The more eco-points a product has the worse it is for the environment\*.

A different approach has been developed in Sweden, although the outcome is the same. In this approach resources are aggregated on the basis of their reserves and the "uniqueness" of the resource, and substances are aggregated using an index for each substance which is based on six aspects. These aspects are:

- type and gravity of the problem which the intervention contributes to;
- intensity and frequency with which the problem occurs;
- spatial distribution of the problem;
- duration of the problem;
- contribution to the problem per kg emissions;
- options for and costs of emission reductions.

If a substance contributes to more than one problem an index is calculated for each type of problem and these indices are added to provide the total index of that substance. The index is then multiplied by the emissions and the results for all emissions are totalled. Again, the higher the total, the worse the product.

The method based on units of polluted air and water (see §3.1.3), sometimes known as the *critical volumes method*, is also based on a combination of classification and evaluation. According to this method substance emissions into the atmosphere are divided by the MAC value (*maximum accepted concentration* based on workplace exposure) and emissions into water are divided by the value for surface water intended for the preparation of drinking water (OVD). In this way a volume of polluted air or water can be calculated. These volumes are added together for all emissions by type of air or water. The MAC and OVD values used are partly based on societal (i.e. political, economic and technical) feasibility and not exclusively on environmental aspects.

Although all three approaches provide eminently practical results they have been subject to severe criticism, mainly because of the combination of societal feasibility and environmental aspects. In this study we will therefore attempt to separate these aspects wherever possible and the scientific and social science aspects will be considered in different components of the method: the classification and the evaluation. In this chapter we will focus on the development of the classification, while the development of the evaluation will be discussed in Chapter 4.

## 3.1 General principles

The general definition, premises and mathematical structure of the classification will be examined in this section. It also contains a list of environmental problems which are included in the method.

### 3.1.1 Definition of the classification

In view of the introduction the classification, as a component in an LCA, can be defined as a quantification of the environmental interventions of a product system:

- with regard to a number of commonly accepted environmental problems;

\* Note that an eco-point has the dimension time, see also §4.1.2.

- by problem type;
- while considering all relevant environmental processes.

Generally the volume of data will be reduced by the classification as the lists of dozens to hundreds of different types of environmental interventions, are aggregated to between ten and twenty effect scores. However, this reduction is not the main objective. In principle it is possible that a list of, for example, five environmental interventions is expanded to seven effect scores in the classification, as the interventions are quantified by all their potential effects. The term potential will be discussed below (see §3.1.4).

Possible classification methods were discussed at a workshop in Florida in February 1992. These discussions resulted in a list of four methods which could assist in the classification of substance emissions:

- *loading analysis*; aquatic emissions are added without weighting and atmospheric emissions are added without weighting (Hunt *et al.*, 1974);
- *equivalent unit analysis*; emissions are added on the basis of their potential effects without consideration of environmental processes and exposure routes;
- *generic exposure effect analysis*; emissions are added on the basis of a generic analysis of environmental processes, exposure routes and potential effects, while taking into account the average background concentration;
- *site specific exposure effect analysis*; emissions are added on the basis of a site specific analysis of environmental processes, exposure routes and potential effects, while taking into account the background concentrations on the site concerned.

Although this suggests that there are many options for the classification a further consideration of the four methods shows that this is not so.

*Loading analysis*, for example, is a method which does not fulfil any of the elements of the definition of the classification. This method applies the principle "less of any intervention is better" but does not allow a meaningful comparison between interventions of different types.

In the context of an LCA, which concerns dozens of processes which may occur anywhere in the world, a *site specific exposure effect analysis* is impossible. This method is rather more compatible with an environmental impact assessment when an environmental analysis has to be made of one plant or activity at a specified site.

In *equivalent unit analysis* the potential effects of emissions are studied without considering previous environmental processes. Examples of this approach include the critical volumes method for the classification of acidifying and nitrifying substances (see below).

In a *generic exposure effect analysis* the potential effects of an emission are appraised on the basis of a generic analysis of environmental processes and exposure routes. Examples of this form of analysis include the classification of greenhouse gasses on the basis of their *global warming potential* (GWP; §3.2.3), the classification of substances which deplete the ozone layer on the basis of their *ozone depletion potential* (ODP; §3.2.4) and the classification of substances which form photochemical oxidants on the basis of their *photochemical ozone creation potential* (POCP; §3.2.7). In an LCA the aim is to use generic exposure effect analysis for substance emissions. Equivalent unit analysis may provide a provisional solution until a method is available in which environmental processes and exposure routes are taken into account.

Four steps can be distinguished in the development of the classification based on an equivalent unit analysis:

- selection of the problem types (step 3.1);
- definition of the classification factors (step 3.2);
- creation of the environmental profile (step 3.3)
- normalization of the effect scores (step 3.4).

The nature of the first two steps is more general than that of the last two. This report includes a proposal for the first two steps to provide a *standard model* for practical studies. However, a different version of this standard model can be used, for example because it does not meet all the requirements of a given study. In this case the first and second steps will also be used in practice. The third and fourth steps are only relevant when the LCA is carried out in practice. In the following we will



concentrate on the methodological aspects of the steps.

The selection of problem types and definition of the classification factors will be discussed in separate sections. The methodological aspects of the last two steps will be discussed together in the section about the definition of the classification factors.

### 3.1.2 Spatial differentiation

How to deal with spatial differentiation is a major issue in the development of classification factors for each problem type. Spatial differentiation may be relevant, for example, when the degradation or biological availability of a substance depends on the soil type. For example, the biological availability of toxic substances in the soil depends on the fraction of humus particles and the fraction of organic matter. The harmful effects of acid deposition on a given area depend on the buffer capacity of the trees and type of ecosystem in the area (coniferous forest, deciduous forest, heather).

Spatial differentiation could be based on four levels of scale: global, continental, regional and local. Two approaches could then be taken on each level of scale:

- a *location specific approach* in which relevant parameters for a given level of scale are localized on a geographical map and emissions are converted to concentrations at specific locations with the aid of a dispersal-deposition model;
- a *generic approach* in which averaged relevant parameters are specified in percentages for a given level of scale and a range of location types.

The RAINS acidification model (Alcamo *et al.*, 1987) developed by the IIASA (International Institute of Applied Systems Analysis) is an example of a location specific approach. In this model the effects of acid rain deposition are studied in relation to the geographical distribution of sensitive, unbuffered areas throughout Europe while the background concentrations are also taken into account.

Some authors appear to favour the use of such a detailed approach in LCAs (Grienshammer *et al.*, 1991; Fava *et al.*, 1991). Such an approach, however, demands a lot of the data provided by the inventory analysis and the classification. In the inventory analysis the geographical location of economic processes would have to be specified while data about the abiotic and biotic structure of the areas around the site of each process would have to be collected in the classification. Such an approach does not appear to be feasible in the context of an LCA in which dozens or even hundreds of processes occur at as many sites throughout the world.

Hence, a generic approach would appear to be more feasible in an LCA. However, even when a generic approach is adopted the requirements made of the data from the inventory analysis and the classification may differ greatly for each level of scale. More and more data will be needed as the focus changes from a generic approach at global level to a generic approach at local level. As the world is divided into smaller and smaller scale units the classification of each unit (the five continents, thousands of regions and millions of locations) would have to contain the average data for the relevant parameters. During the inventory analysis the processes would also have to be specified on this scale of units. Clearly there will be little difference between a generic approach at the local scale and a detailed approach at the local scale so it will be difficult to distinguish these two approaches at this level of scale.

This report provides a first step towards the generic development of classification factors at the global level. Hence classification factors are not developed for different areas. If necessary a single "world average soil composition" will be used or the "soil composition representative of the world". Wherever possible the way in which a generic approach at a lower scale level could be developed will be indicated for problem types where spatial differentiation is relevant.

The following assumptions were used in the generic development of the classification factors:

- classification factors will not be specified in spatial terms, even when spatially differentiated data is available in the inventory analysis;
- any background concentrations relevant to classification factors will only be included as "world average" or "representative for the world".

### 3.1.3 Mathematical background

The following three mathematical subjects will be discussed in this section: the use of classification factors as a linear approach to the world model, the nature of environmental models and the historical

confusion about the nomenclature of parameters and units in the classification.

#### The classification factor as a linear approach

During the classification environmental models are used to translate environmental interventions into potential environmental effects. In mathematical terms an environmental model can be presented as a function ( $f$ ) which maps an environmental intervention ( $x$ ) onto a potential environmental effect ( $y$ ). The mapping of intervention  $x$  onto effect  $y$  is represented by

$$f: x \rightarrow y \quad (3.3)$$

and the functional nature of the model  $f$  as

$$y = f(x) \quad (3.4)$$

During the discussion of the premises (§0.4) and the process data (§2.2.1) it was remarked that it is assumed that there is a *ceteris paribus* situation and that the interventions and effects caused by the product system are small compared with the overall problem\*. During the inventory analysis of the process data this resulted in the definition of the principle that, ideally, long term marginal process data would have to be collected. During the classification using models this resulted in the linearisation of the models. When the total of a given intervention  $x$  throughout the world is increased by the marginal intervention  $\Delta x$  the total worldwide effect  $y$  will be increased by a quantity  $\Delta y$ :

$$y + \Delta y = f(x + \Delta x) \quad (3.5)$$

This is equal to

$$\Delta y = f(x + \Delta x) - f(x) \quad (3.6)$$

as  $\Delta x \ll x$  this may be linearised to

$$\Delta y = \left[ \frac{\partial f}{\partial x} \right]_x \times \Delta x \quad (3.7)$$

where  $[\partial f / \partial x]_x$  represents the slope of the tangent of the function  $f$  at  $x$  (see Figure 3.1). The linearity is illustrated even more clearly when the linearised model is presented as a model factor  $c$ :

$$\Delta y = c \times \Delta x \quad (3.8)$$

\* These conditions may not apply when LCAs are used for scenario studies: the interventions may be significant by comparison with the total when modelling scenarios to compare traffic situations in a country. This may introduce an error in the predicted effect depending on the extent of non-linearity.

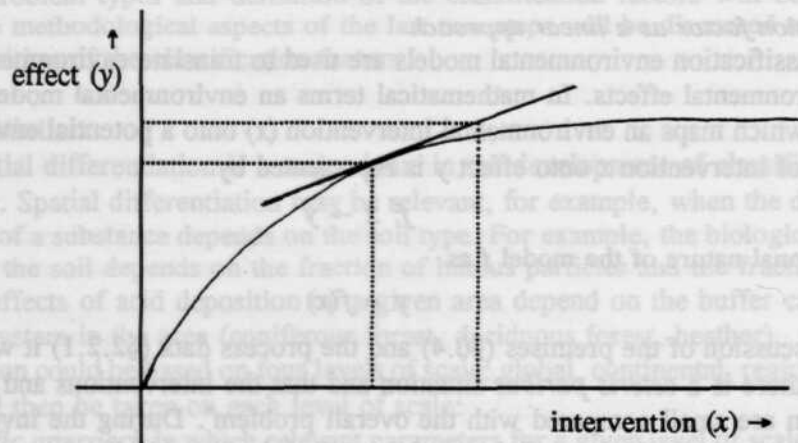


FIGURE 3.1. The interventions are assumed to be marginal compared with the background concentrations. Hence the local tangent of the non-linear intervention-effect equation can be used as a classification factor.

It is thus "demonstrated" that classification factors can be used in a generic classification of environmental problems. Here classification factors are used in the literal sense, i.e. an the potential effect is represented by an intervention multiplied by a factor\*.

#### Nature of the environmental models

There are two types of model used for the classification:

- mechanism-oriented models;
- effect-oriented models.

These types will be discussed in general mathematical terms using an example.

#### Mechanism-oriented models: the equivalence factor

The model use to classify greenhouse gas emissions is an example of a *mechanism-oriented model*. The eventual effect (in this case global warming, or even further down the effect chain, the rise in the sea level and damage to ecosystems; see Figure 3.3) is not included in the model. Instead the mechanism of the effects of a reference substance are compared with the effects of the other substances under the same conditions. A relative measure is then developed on this basis which indicates the contribution to the mechanism of a standard quantity of that substance.

The *global warming potential* (GWP) of a substance  $i$  is defined as:

$$GWP_i = \frac{\int_0^T a_i c_i(t) dt}{\int_0^T a_{ref} c_{ref}(t) dt} \quad (3.9)$$

where  $T$  is the time horizon,  $c_i(t)$  the concentration of substance  $i$  at time  $t$ ,  $a_i$  the effect of a mass unit of substance  $i$  and the symbols marked *ref* represent the corresponding parameters for the reference substance (Houghton *et al.*, 1991). A time horizon  $T$  of 100 years and the use of CO<sub>2</sub> as the reference substance *ref* are most commonly used. This definition clearly illustrates that we are interested in the

\* It is assumed that the models used for the classification include any non-linearity in the dose-response relationship or that the models are linear but valid in a limited linearised domain. The models used to classify greenhouse gasses meet this requirement. However for the effects of toxic substances further research is needed into the validity of the critical volume approach used here in which a threshold (NOEC, ADI) is used as a measure of the toxic effect.



enhancement of the greenhouse effect of a substance  $i$  compared to a reference substance  $ref$ . This also shows that the GWP is a dimensionless unit.

A parameter with the unit mass is obtained by multiplying the emitted quantity (expressed in mass units) of substance  $i$  by the GWP of that substance. This can be interpreted as the emission of the reference substance  $ref$  which enhances the greenhouse effect to the same extent as the emissions of  $i$ . The parameter determined in this way could be called the "emission equivalent to  $CO_2$  in terms of the greenhouse effect with a time horizon  $T$  of 100 years". As this name is rather cumbersome the term "greenhouse effect" is used, with the implicit agreement that  $CO_2$  is used as the reference substance  $ref$  and that the time horizon  $T$  is 100 years. Like any other mass in the SI system the greenhouse effect is expressed in kilograms (kg).

The general form of the calculations made with a mechanism-oriented model in the classification is as follows:

$$\Delta y = \text{equivalence factor} \times \Delta x \quad (3.10)$$

The equivalence factor is dimensionless, hence the effect  $y$  has the same dimension as intervention  $x$ . In the example above:

$$\text{equivalence factor} = \text{GWP} \quad (3.11)$$

#### *Effect-oriented models: the conversion factor*

In many studies an *effect-oriented approach* is used for the classification of toxicity when inhaled, rather than a mechanism-oriented one. The critical volumes approach previously used by CML will be used as an example. A *maximum accepted concentration* (MAC) can be determined for each substance, based on toxicological data and other information. The MAC is defined as the highest concentration whose observed effects on man are considered just acceptable (Arbeidsinspectie, 1989). In this case the effect, rather than the mechanism causing that effect, is considered.

A parameter with the dimension volume is obtained by dividing the quantity of a substance emitted into the atmosphere by the MAC of that substance. This parameter can be interpreted as the volume of air contaminated by that substance up to the MAC. Thus this parameter could be called "volume of air contaminated up to the MAC", which would be another impossibly long name. In the past the name UPA was chosen. The implicit agreement is that the MAC according to the latest list published by the Netherlands Labour Inspectorate is used. Like any other volume this parameter is expressed in  $m^3$ . However there is some confusion about nomenclature which is discussed below.

Calculations made with effect-oriented models in the classification take the following general form:

$$\Delta y = \text{conversion factor} \times \Delta x \quad (3.12)$$

In most cases the conversion factor has a unit, hence the effect has a different dimension than the intervention. In the example above:

$$\text{conversion factor} = \frac{1}{\text{MAC}} \quad (3.13)$$

As the MAC is a concentration expressed in  $mg \cdot m^{-3}$  the unit of the conversion factor is  $m^3 \cdot mg^{-1}$ . The intervention is an emission, expressed in mg. This implies that the effect is expressed in  $m^3$  and corresponds with a volume, hence the term *critical volume*.

#### *Unifying the models: the classification factor*

Both mechanism-oriented and effect-oriented models can be used in the classification. The term *classification factor* will be used as a unifying term for the equivalence factor and the conversion factor:

$$\Delta y = \text{classification factor} \times \Delta x \quad (3.14)$$

$\Delta x$  is the magnitude of the intervention of a functional unit;  $\Delta y$  is the contribution to a given environmental problem: the effect score. Whether the classification factor has a dimension or not depends on the effect score. To make the notation more convenient and to ensure uniformity with the notation used in the inventory analysis (§2.4) any environmental intervention  $\Delta x$  will be indicated as

$\beta_k$  and an effect score  $\Delta y$  with  $\gamma_i$ . The effect score is calculated with

$$\gamma_i = \sum_{k=1}^s c_{ik} \beta_k \quad (3.15)$$

where  $c_{ik}$  is the classification factor linking the  $k^{\text{th}}$  environmental intervention with the  $i^{\text{th}}$  effect score.

### Quantities and units

The foundations for classification as a part of life cycle assessment were laid in several places in the past. Owing to this geographical distribution there is currently some confusion about the meaning of certain terms. The aim of this section is to define clear choices within the framework of a standardized method. This largely concerns quantities describing environmental effects and the units in which they are expressed.

So far the MAC list has generally been used for the classification of air pollution: the emissions of a substance were divided by the MAC value of that substance to obtain "units of polluted air". However, the definition of UPA is not uniform. Some studies do not define it explicitly. The following definitions, either explicit or implicit, are used:

- UPA is a unit: 1 UPA = 1000 m<sup>3</sup> polluted air (Druijff, 1984);
- UPA is a unit: 1 UPA = 1 m<sup>3</sup> polluted air (Muis *et al.*, 1989);
- UPA is a quantity expressed in m<sup>3</sup> (Rijsdorp *et al.*, 1989).

One report has three different interpretations of manure equivalents (MEQ) for the effect score nitrification:

- MEQ is a unit: 1 MEQ = 1000 kg phosphorus;
- MEQ is a unit expressed in kg;
- MEQ is an equivalence factor.

There are some problems with the use of EVL or MEQ as a unit because there is an international committee to approve new units (Klein, 1988). Units should not be defined outside this system. The following are relevant when admissibility is considered:

- reducing long expressions (1 N = 1 kg·m·s<sup>-2</sup>);
- avoiding prefixes (1 Å = 10<sup>-10</sup> m);
- preference for obvious measures (1 light year = 9.461·10<sup>15</sup> m).

This does not apply to the UPA or the MEQ.

The semantic disadvantage of the use of the UPA as a quantity is that it is a quantity but called a unit. This is one of the reasons why the use of the term UPA is avoided in the development of the human toxicity effect score.

### 3.1.4 The classification factor

The emission classification factor of substances can be considered as consisting of two elements: the *exposure factor* and the *effect factor*. The exposure factor indicates what quantity of a given receptor may be exposed as a result of the emission of a certain substance. The effect factor identifies the effects in a receptor which a given exposure may lead to. These factors are illustrated in Figure 3.2. Both factors are determined for each substance individually and based on scientific models and empirical data where available. The models used to simulate the fate of a substance in the environment provide the basis for the exposure factor. For the ecotoxicity, for example, the effect factor is derived from toxicity data obtained from experiments, such as the *no observed effect concentration* (NOEC). The general definition of the exposure and effect factors is defined for each problem type while the actual values are determined for each substance.

In the development of the classification factors for environmental problems associated with substance emissions we observe that some classification factors can be typified as exposure factors, some as a mixture of exposure and effect factors and some that can be divided into an exposure factor and an effect factor. In theory there is a clear distinction between the two types of factor while in practice, for some substances both factors have been combined into a single index.

Emitted substances can contribute to environmental problems in three ways:

- *in parallel*, the emission of a substance may potentially contribute to several problems;
- *directly in series*, the emission of a substance may have several effects, one after another;



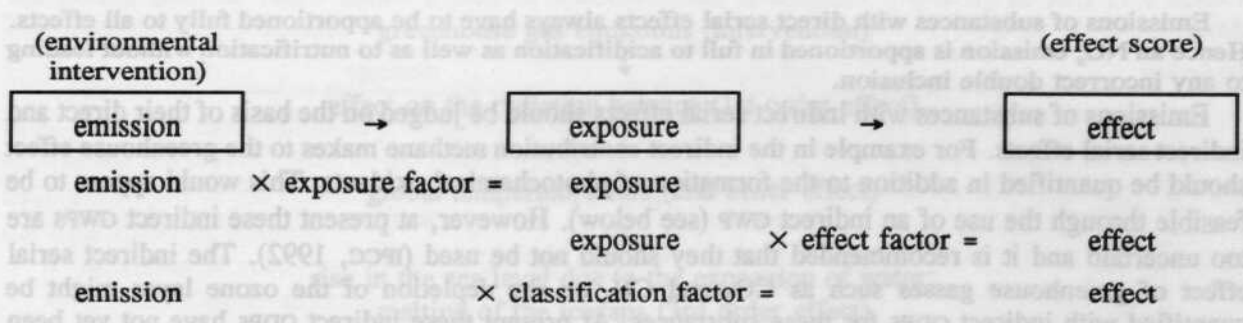


FIGURE 3.2. The relationship between emission, exposure and effect and the roles of the emission, effect and classification factors.

- *indirectly in series*, the emission of a substance contributes to another effect through the effect which it primarily causes.

SO<sub>2</sub> is an example of a substance with parallel effects. This substance can contribute to the acidification of the environment and to toxic effects. However a single molecule of SO<sub>2</sub> cannot contribute to both problems during its existence. Essentially this applies to all non-persistent substances. A single molecule of such a substance can contribute to human toxicity or to ecotoxicity but generally not to both problems concurrently.

Nitrogen oxide (NO<sub>x</sub>) provides an example of a substance with *direct serial* effects. A single molecule of NO<sub>x</sub> can contribute to both acidification and eutrophication. Highly persistent substances such as metals are other examples of substances with direct serial effects. These substances may have an ecotoxic effect or, via the food chain, a human toxic effect through fish consumption for example.

Methane (CH<sub>4</sub>) is an example of a substance with *indirect serial* effects. If the temperature of the troposphere rises due to the greenhouse effect the temperature of the stratosphere will fall, as a result of which the efficiency of the chemical processes which deplete the ozone layer will increase or decrease. At present it is not yet clear how the cooling of the stratosphere affects these chemical processes. One molecule of methane can contribute to the formation of photochemical oxidants and produce ozone which can then contribute to the greenhouse effect. Through the greenhouse effect the same molecule of CH<sub>4</sub> can also contribute to the depletion of the ozone layer. In principle, the distinction between parallel, directly serial and indirectly serial effects should be considered during the classification of substance emissions.

Preferably substance emissions with parallel, directly serial and indirectly serial effects should be attributed to the environmental problems concerned in proportion to their actual contributions. However, not enough is known about these proportions. For this reason all the potential contributions of an emission to the various environmental problems are quantified on the basis of the total quantity emitted. In case of an emission of 2 kg SO<sub>2</sub>, for example, the contribution to both acidification and human toxicity the full 2 kg are quantified. This may lead to some "double inclusion". However, it can be demonstrated that the extent of this double inclusion is negligible for substances with human toxic effects and other parallel effects\*.

\* The proportion of an emission absorbed by man has to be known to be able to estimate the extent of double inclusion. Human absorption depends on location; a larger part of an emission in a residential area will be absorbed by man than an emission at sea. The estimate below aims not to underestimate the quantity absorbed by man.

Assuming a single emission of  $x$  kg toxic substance T to the air and that this substance is not degraded, that it is not taken up by other environmental media, that it is homogeneously distributed through the air and that the atmosphere has a volume of  $3 \cdot 10^{18}$  m<sup>3</sup> (Mackay, 1991), the concentration of T in the air will be  $x/(3 \cdot 10^{18})$  kg·m<sup>-3</sup>. People breathe 20 m<sup>3</sup> air per day. As the current world population numbers  $5 \cdot 10^9$  a total of  $20 \times 5 \cdot 10^9 = 10^{11}$  m<sup>3</sup> air is inhaled per day. Assuming an average lifespan of 100 years (*overestimate*) and that the inhaled quantity of T is fully absorbed by the body (another clear *overestimate*) the quantity absorbed by the entire world population in 100 years amounts to  $100 \times 365 \times 10^{11} \times x/(3 \cdot 10^{18}) = 1.2 \cdot 10^{-3} \times x$  kg T "taken up", i.e. about 1‰ of the quantity originally present. Even this is a clear overestimate of the actual quantity taken up. Hence it appears that the double inclusion of substances with both human toxic



Emissions of substances with direct serial effects always have to be apportioned fully to all effects. Hence an  $\text{NO}_x$  emission is apportioned in full to acidification as well as to nutrification without leading to any incorrect double inclusion.

Emissions of substances with indirect serial effects should be judged on the basis of their direct and indirect serial effects. For example in the indirect contribution methane makes to the greenhouse effect should be quantified in addition to the formation of photochemical oxidants. This would appear to be feasible through the use of an indirect GWP (see below). However, at present these indirect GWPs are too uncertain and it is recommended that they should not be used (IPCC, 1992). The indirect serial effect of greenhouse gasses such as  $\text{CO}_2$  and  $\text{CH}_4$  on the depletion of the ozone layer might be quantified with indirect ODPS for these substances. At present these indirect ODPS have not yet been established. Finnveden *et al.* (1992) suggest that substances with indirect serial effects should be included separately in the classification and that the total emissions of each substance should be listed (in kg), until indirect classification factors for these substances are available. The total score of the substances whose classification factors are known should also be given. Finnveden *et al.* also suggest that substances be included in subscores in the same way whose classification factors are unavailable but which are known to contribute to a given problem (e.g.  $\text{NO}_x$  and NO for which the ODPS are unavailable and  $\text{NO}_x$  for which there is no POCP). This would result in five subscores for the greenhouse effect: one for substances with GWPs (in kg  $\text{CO}_2$  equivalents), one for the total  $\text{CH}_4$  emissions (in kg  $\text{CH}_4$ ), one for the total CO emissions (in kg CO), one for the total  $\text{NO}_x$  emissions (in kg  $\text{NO}_x$ ) and one for the total emissions of non-methane hydrocarbons (NMHC, in kg NMHC). There would be five subscores for the depletion of the ozone layer: one for the CFCs, excluding halons, based on their ODP values (in kg CFC-11 equivalents), one for halons based on their ODPS (in kg halon-1301 equivalents), one for the total  $\text{N}_2\text{O}$  emissions (in kg  $\text{N}_2\text{O}$ ), one for the total  $\text{CO}_2$  (in kg  $\text{CO}_2$ ), one for the total  $\text{CH}_4$  emissions (in kg  $\text{CH}_4$ ) and one for the total NO emissions by aeroplanes (in kg NO). This proposal leads to a large number of subscores with widely varying statuses. Hence it would appear to be better and more practical to deal with these uncertainties as such. For example "flags" (qualitative remarks) could be attached to substance emissions which may have indirect effects. The values of the associated indirect GWPs and ODPS at which the outcome of the LCA would change (see §4.2) could then be calculated in a reliability analysis. It could then be considered whether the values calculated are realistic, given the current level of understanding. Substances known to contribute to certain problems but for which classification factors cannot yet be determined could be dealt with in a similar manner.

To summarize: until the actual proportions are known, the emissions of substances with parallel effects as well as the emissions of substances with direct serial effects are apportioned in full to each relevant environmental problem. For substances with parallel effects this may lead to some double inclusion. However for toxic substances this appears to be marginal. Emissions of substances with indirect serial effects should be marked with "flags" until there are classification factors for these indirect effects. The effect of this uncertainty on the endresult can then be estimated during the reliability analysis.

### 3.1.5 Definition of the environmental problems

The first element in the classification is the definition of a list of commonly recognized environmental problems which should be covered by an LCA. Environmental problems may be formulated either in terms of processes/mechanisms or in terms of environmental effects. Both methods may refer to different parts of the chain of environmental effects. The effect chain of the greenhouse effect is shown as an example in Figure 3.3.

The greenhouse effect is the result of the emission of a range of substances. These substances absorb infrared (heat) radiation which disturbs the heat balance of the heat radiation absorbed by earth and the heat radiation emitted by earth. This change in the radiant properties of the atmosphere is known as the greenhouse effect, it is the primary effect in the chain of effects. According to the predictions this change will result in a global temperature rise (secondary effect) which will have

and other effects is negligible.

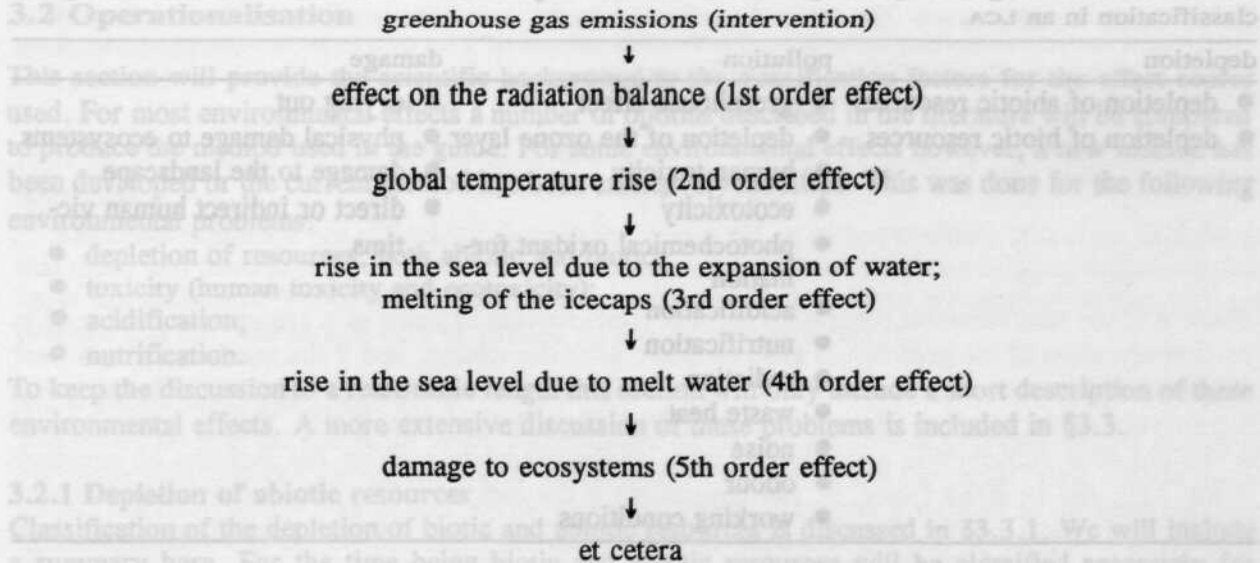


FIGURE 3.3. Part of the environmental effect chain for the greenhouse effect (see also Finnveden *et al.*, 1992).

further effects (tertiary, quaternary, etc.) on the icecaps, the sea level, ecosystems, etc. (Finnveden *et al.*, 1992). Many forms of feedback can occur within an effect chain, as well as between effect chains, as referred to above under parallel and serial effects.

Generally the predictability of the effects in the chain decreases as the order of the effects increases. In principle changes should therefore be linked to the lowest order effect which shows a clear relationship with the effect chain under consideration. For example, the ozone layer or our climate are selected as an end point in the classification, although they are only steps in the chain of environmental effects\*.

The first international discussion about the environmental problems which should be considered in an LCA was held during a workshop on LCAs in Leiden in December 1991 (Baumann *et al.*, 1992; Guinée, 1992). An exhaustive list of generally recognized environmental problems was compiled, classified in three main categories:

- *depletion*, including all the problems associated with extraction from the environment;
- *pollution*, including all the problems associated with all types of emissions to the environment;
- *damage*, including all the problems associated with changes in the structure of the environment.

This list is given here with some changes and additions. It includes the types of problems listed in Table 3.1. Please note that all environmental effects have been defined as problems, hence the principle of "more is worse" applies. For example, we are concerned with "lack of safety" rather than with "safety", the problem here is the occurrence of accidents resulting in the effect "victims".

At least three common environmental problems have not been included in this list: energy consumption, solid waste and space use. Energy consumption is not a problem as such but is partly a problem of depletion (depletion of energy carriers), partly a pollution problem (e.g. emission of acidifying substances) and partly a problem of damage (e.g. due to mining). Similarly, solid waste is not a problem as such, but rather a process (storage of solid waste) which can cause emissions to water, the atmosphere and soil and can produce methane as a possible energy source. Space use was also omitted from the list of environmental problems. Naturally the volume of space is limited and in this respect it could be included under depletion, however this is not so much an environmental problem as one of spatial planning. The nature of space use, in terms of damage to and displacement

\* Such a choice of end points in the chain can only be made if the significance attached to them by society is clear. Hence there should be consensus about the significance of the end point. Many environmental problems have already been defined in terms of such mechanisms, some examples were given above.

TABLE 3.1. List of generally recognized environmental problems which could be used during the classification in an LCA.

depletion	pollution	damage
<ul style="list-style-type: none"> <li>● depletion of abiotic resources</li> <li>● depletion of biotic resources</li> </ul>	<ul style="list-style-type: none"> <li>● greenhouse effect</li> <li>● depletion of the ozone layer</li> <li>● human toxicity</li> <li>● ecotoxicity</li> <li>● photochemical oxidant formation</li> <li>● acidification</li> <li>● eutrophication</li> <li>● radiation</li> <li>● waste heat</li> <li>● noise</li> <li>● odour</li> <li>● working conditions</li> </ul>	<ul style="list-style-type: none"> <li>● drying out</li> <li>● physical damage to ecosystems</li> <li>● damage to the landscape</li> <li>● direct or indirect human victims</li> </ul>

of ecosystems is much more important. This aspect will be classified as physical damage to ecosystems.

Other problem types have not been included in the list as it is difficult to attribute them to the functioning of products (e.g. fragmentation of natural areas or depletion of the gene bank) or because, for the present at least, there are no generally recognized environmental problems (e.g. light which may lead to local environmental problems around greenhouses, and growing salinity which is another localized problem). It is therefore likely that the list is incomplete but it could also be extended at a later stage if required. When drawing up this list every effort was made to be consistent with the terms used in the NEPP (VROM, 1989) and the "National Environmental Outlook" (RIVM, 1991).

Classification based on the above list of environmental problems will result in a maximum of 18 environmental problems, some of which consist of more than one element. Whether this maximum number will be used in practice will depend on the availability or development of classification factors for these problems and the data for the inventory analysis\*. We will now consider the availability of classification factors per problem type and the availability of the necessary data. This information will then be used to create a list of problem types whose classification is currently available. This list is included in the guide.

### 3.1.6 The environmental profile

The inventory table for a functional unit of a product lists the environmental interventions associated with it. A table of potential environmental effects, in the form of effect scores, is obtained by sorting, weighting and adding all the environmental interventions in a suitable way. This table is the environmental profile. The environmental profile can be drawn up on the basis of the inventory table (see §2.4), by process or group of processes, substance or group of substances or for a single product. An environmental profile for a single product is required for product policy. The evaluation component is based on the environmental profile. There are various procedures for converting interventions to effects, all of which are based on different assumptions. To some extent the choice made is debatable and is therefore specifically not incorporated in the inventory analysis.

\* Furthermore effect scores may be excluded when the inventory analysis shows that the problem concerned is not relevant in a given product study. For example if the inventory analysis shows that there are no emissions of volatile organic compounds the score for the formation of photochemical oxidants will be zero.



### 3.2 Operationalisation

This section will provide the scientific background to the classification factors for the effect scores used. For most environmental effects a number of options described in the literature will be compared to produce the method used in the guide. For some environmental effects however, a new method has been developed or the current method has been extensively modified. This was done for the following environmental problems:

- depletion of resources, both abiotic and biotic
- toxicity (human toxicity and ecotoxicity);
- acidification;
- eutrophication.

To keep the discussion to a reasonable length this section will only include a short description of these environmental effects. A more extensive discussion of these problems is included in §3.3.

#### 3.2.1 Depletion of abiotic resources

Classification of the depletion of biotic and abiotic resources is discussed in §3.3.1. We will include a summary here. For the time being biotic and abiotic resources will be classified separately for practical reasons. For biotic resources the net annual production (the difference between consumption and growth in one year) has to be included as a measure of the overexploitation (see §3.2.2). This need is not as pressing for abiotic resources. A simplified method will be used as the classification factor is sensitive to the extreme variability of data about the reserves. Eventually the aim will be to use the same method for abiotic resources as for biotic resources.

The quantity of a resource used  $i$  ( $m_i$ ) is linked to the recoverable reserves of that resource ( $M_i$ ):

$$\text{abiotic depletion} = \sum_i \frac{m_i}{M_i} \quad (3.16)$$

As the effect score obtained is dimensionless the dimensions of the consumption and reserves can be selected freely, as long as they are the same for a given resource. Ores will normally be expressed in kg and natural gas in m<sup>3</sup>, although MJ is also possible. Energy carriers will be included depending on whether they are biotic or abiotic; fossil fuels are considered abiotic.

#### 3.2.2 Depletion of biotic resources

The complete method assessing the use of resources will be used for biotic resources (see §3.3.1 for further details). One of the reasons for this is that it is clearly necessary to include the extent of overexploitation when considering biotic resources.

The consumption ( $m_i$ ) of a resource  $i$  is linked to the recoverable reserves ( $M_i$ ) and the annual production  $\dot{M}_i$  of those reserves:

$$\text{biotic depletion} = \sum_i \frac{\dot{M}_i}{M_i} \times \frac{m_i}{M_i} = \sum_i BDF_i \times m_i \quad (3.17)$$

The *biotic depletion factor* (BDF) is then defined as:

$$BDF_i = \frac{\dot{M}_i}{M_i^2} \quad (3.18)$$

Thus the unit of the BDF is kg<sup>-1</sup>·yr<sup>-1</sup>, hence the dimension of the biotic depletion effect score is yr<sup>-1</sup>. Other units are sometimes used, e.g. ha for rain forest and number of individuals for elephants. Thus the BDF will have a different unit but this will not affect the effect score.

#### 3.2.3 Enhanced greenhouse effect

Models have been developed to quantify the contribution made by emissions of various substances to the greenhouse effect. Generally these models provide an indication of the change in the heat radiation absorption (*radiative forcing*) of the atmosphere. This is popularly referred to as the "greenhouse effect". *Global warming potentials* (GWPs) have been developed to compare the emission scenarios of

different greenhouse gasses. The GWP of a substance is the ratio between the contribution to the heat radiation absorption resulting from the instantaneous release of 1 kg of a greenhouse gas and an equal emission of carbon dioxide (CO<sub>2</sub>) integrated over time:

$$GWP_i = \frac{\int_0^T a_i c_i(t) dt}{\int_0^T a_{CO_2} c_{CO_2}(t) dt} \quad (3.19)$$

where  $a_i$  is the heat radiation absorption per unit concentration increase of a greenhouse gas  $i$ ,  $c_i(t)$  is the concentration of the greenhouse gas  $i$  at time  $t$  after the release, and  $T$  the number of years over which the integration was made. The corresponding values for carbon dioxide are included in the denominator of the equation (Houghton *et al.*, 1991; Jäger & Ferguson, 1991). The GWP is a measure of the potential contribution a substance makes to the greenhouse effect and includes an exposure factor,  $c_i(t)$ , and an effect factor,  $a_i$ , the atmosphere is the receptor\*. The GWP is only an estimate of the eventual effect on the climate, as this depends not only on the integrated atmospheric heat absorption but also on the distribution over time of the heat absorption.

A number of simplifications are made when integrating the greenhouse effect. The GWPs, for example, depend on the time horizon  $T$  over which the integration is made. Longer time horizons (100 and 500 years) are used for the cumulative effect, while shorter time horizons (20 and 50 years) provide an indication of the short-term effects of the emissions. Uncertainties in the GWP increase as the time horizon becomes more distant. The reason for this is that GWPs depend on the background concentrations of certain substances in the atmosphere. If these concentrations change the GWPs will also change. For example the life span of substances which are eliminated by OH radicals in the atmosphere may change significantly, depending on future anthropogenic emissions of methane, carbon monoxide and nitrogen oxides (Houghton *et al.*, 1991; Jäger & Ferguson, 1991). Such changes in the lifespan have a major, often disproportional, effect on all the GWPs of the substances concerned as they are defined in relative terms.

Several authors have developed models to calculate GWPs (Fisher *et al.*, 1990; Lashof & Ahuja, 1990; Rodhe, 1990; Derwent, 1990; Rotmans, 1990). As a result different values have been calculated for each substance. The *Intergovernmental Panel on Climate Change* (IPCC) has compiled a list of "provisional best estimates" for GWPs based on the expert judgement of scientists worldwide. This list of GWPs will be used here to classify greenhouse gasses because of its broad support.

The integration time to be used will be decided by the user of the GWPs and period over which the effects have to be studied. A long time horizon would appear to be preferable for classification as part of an LCA as we are concerned with all effects, not just the short term ones. However, as referred to above, the uncertainties in the models increase with longer integration times. Hence, it would appear best to use different time horizons and a range of values. Our suggestion is to study the GWP with time horizons of 20, 100 and 500 years (Houghton *et al.*, 1991). In practice this means that an integration time of 100 years is used initially, for example. The range of the result is then determined by undertaking the calculation with the GWPs for the other integration periods as well.

As mentioned above, ozone in the troposphere is also a potential greenhouse gas. Direct ozone emissions due to electrical discharges and high temperatures (e.g. photocopiers) are negligible (Van den Born *et al.*, 1991) compared with indirect ozone formation due to photochemical reactions as described in the section on the photochemical formation of oxidants (§3.2.7). The IPCC has published indirect GWPs for the indirect greenhouse effect of CH<sub>4</sub>, CO, NO<sub>x</sub> emissions and non-methane hydrocarbons (NMHC) emissions (Houghton *et al.*, 1991). However, these were recently retracted in another IPCC report (Houghton *et al.*, 1992). The uncertainties surrounding these values were too great and some substances might even have a positive rather than a negative effect. For the time being the most practical approach appears to be only to use GWPs to quantify the direct effect and to include the

\* This is not quite correct; the matter is slightly more complicated due to the integration and the definition of the GWP relative to a reference substance.



indirect effect as a qualitative item (see §3.2.19). This would produce a single effect score for the greenhouse effect:

$$\text{greenhouse effect} = \sum_i \text{GWP}_i \times m_i \quad (3.20)$$

Here  $m_i$  is the mass (in kg) of the substance released. The effect score is expressed in kg CO<sub>2</sub> equivalents.

Rapid developments are still taking place in the area of the greenhouse effect. It is clear, therefore, that these will have to be closely followed and that GWPs should be modified in accordance with the latest developments. To avoid the need to change the GWPs after every publication in this area we propose that the GWP should only be revised further to new IPCC reports.

### 3.2.4 Depletion of the ozone layer

The ozone layer is the end point in the classification of this environmental problem. *Ozone depletion potentials* (ODPs) have been developed for substances which can deplete the ozone layer, similar to the GWPs. The ODP concept was introduced by Wuebbles (1988). The ODP is defined as the ratio between ozone breakdown in the equilibrium state due to annual emissions (flux in kg·yr<sup>-1</sup>) of a quantity of a substance released into the atmosphere and the breakdown of ozone in the equilibrium state due to an equal quantity of CFC-11 (WMO, 1989):

$$\text{ODP}_i = \frac{\delta[\text{O}_3]_i}{\delta[\text{O}_3]_{\text{CFC-11}}} \quad (3.21)$$

where  $\text{ODP}_i$  represents the ODP of substance  $i$ ,  $\delta[\text{O}_3]_i$  the change in the ozone column in the equilibrium state due to the annual emissions of substance  $i$ , and  $\delta[\text{O}_3]_{\text{CFC-11}}$  the change in the ozone column in the equilibrium state due to the annual emissions of CFC-11. It can be shown that an ODP based on an emission flux (kg·yr<sup>-1</sup>) also provides a good measure of the relative changes in the ozone column due to an instantaneous emission (kg) to the atmosphere (WMO, 1989).

Several models provide ODPs, within a reasonable range, for halogenated hydrocarbons. However, there is still considerable uncertainty about the ODP concept and like GWPs it must be stressed that ODPs will have to be modified regularly to reflect the latest developments. For the same reasons as for GWPs, we propose to adopt the list of ODPs and any future modifications published in WMO (1989) for the classification of substances which can deplete the ozone layer. This report was compiled by the *Scientific Assessment Panel* of which all the authorities in this field are a member. It is therefore reasonable to assume that the results will be widely supported internationally. The major ODP models are discussed in this report and the range of ODPs is specified on the basis of these models (WMO, 1989) which can be used in an LCA classification in the same way as the ranges in GWPs.

ODPs have only been derived for halogenated hydrocarbons. According to the WMO the ODPs of bromine compounds (bromohalogenes) among the halogenated hydrocarbons "can only be compared relative to each other, because of the strong dependence of bromine effects on ozone to background chlorine levels" (WMO, 1989). This has now been superseded by further research (WMO, 1991). Besides halogenated hydrocarbons greenhouse gasses can also affect the ozone column, either directly via chemical reactions (N<sub>2</sub>O and NO<sub>2</sub>) or indirectly through the greenhouse effect (CH<sub>4</sub> and CO<sub>2</sub>; see also §3.3). However, the effects of halogenated hydrocarbon emissions and N<sub>2</sub>O cannot be added. On the one hand an increased N<sub>2</sub>O concentration results in damage to the ozone layer as the NO<sub>x</sub> released by this catalyses the breakdown of ozone. On the other hand there is also an antagonistic effect due to the formation of stratospheric reservoir compounds such as ClONO<sub>2</sub> through the reaction of NO<sub>x</sub> (emitted as N<sub>2</sub>O) and Cl (released by halogenated hydrocarbons) and due to the increased concentration of greenhouse gasses. This results in a temperature drop which increases the ozone column. The complex interactions between the depletion of the ozone layer and the greenhouse effect are still unclear and uncertain. For this reason ODPs have not yet been determined for unhalogenated hydrocarbons (RIVM, 1991; Baumann, 1992).

Therefore it has not yet been possible to determine an ODP for N<sub>2</sub>O. Similarly there is no ODP for NO emissions from planes. We propose that NO and N<sub>2</sub>O should be dealt with in the same way as other greenhouse gasses with a serial ozone depletion effect (CH<sub>4</sub>, CO<sub>2</sub>). Finnveden *et al.* (1992)



however propose to develop a separate score for  $N_2O$ . This score would be equal to the total  $N_2O$  emissions in kg. However, this proposal would have to be implemented for all substances for which an ODP has not yet been set ( $N_2O$ ,  $CO_2$ ,  $CH_4$  and  $NO$  from planes). This would result in five subscores for the depletion of the ozone layer.

According to the method we have proposed above the direct ozone depletion effect can be quantified as a single effect score:

$$\text{ozone depletion} = \sum_i ODP_i \times m_i \quad (3.22)$$

where  $m_i$  is the emission in kg and the effect score is expressed in kg (CFC-11 equivalents).

### 3.2.5 Human toxicity

Man forms the end point in the classification of human toxicity. Although separate classification factors have been defined for the environmental media air, water and soil the effect scores for these media can be added without weighting to provide a single medium-independent effect score for human toxicity.

The *exposure factor*  $X$  (see §3.1.4) is divided into three parts: a factor  $X_a$  for direct exposure to emissions to the air, a factor  $X_w$  for direct exposure to emissions into surface waters and a factor  $X_s$  for direct and indirect exposure to emissions into the soil. Similarly the *effect factor*  $E$  is divided into  $E_a$ ,  $E_w$  and  $E_s$  for effects due to direct intake through air or water and direct and indirect intake through the soil. These effect factors are calculated by using the ADI (acceptable daily intake) values defined by the WHO and the TDI (tolerable daily intake; Vermeire *et al.*, 1991) values similarly defined by the RIVM. Furthermore the human toxicological *maximum tolerable risk level* (MTR; VROM, 1988a) is used for carcinogenic genotoxic substances. Wherever possible the TCL (acceptable concentration in air; Vermeire *et al.*, 1991) values or the *air quality guidelines* (WHO, 1987) are used to determine the effect factor for air.

Separate classification factors are calculated for each of the media air, water and soil. The *human toxicological classification factors for air, water and soil* ( $HCA$ ,  $HCW$  and  $HCS$ ) are defined as

$$HCA = B_a \times E_a \quad (3.23)$$

$$HCW = B_w \times E_w \quad (3.24)$$

and

$$HCS = B_s \times E_s \quad (3.25)$$

The way in which the various exposure factors and effect factors are calculated depends on the basic data available. The method used to calculate these factors for this report is discussed in §3.3.3.

In a product study the classification factors for air, water and soil can be used to calculate a single effect score for human toxicity for each product, independent of the medium. Based on the emissions  $m_{a,i}$ ,  $m_{w,i}$  and  $m_{s,i}$  (all in kg) to the atmosphere, water and soil, the effect score for human toxicity (in kg body weight) is

$$\text{human toxicity} = \sum_i ((HCA_i \times m_{a,i}) + (HCW_i \times m_{w,i}) + (HCS_i \times m_{s,i})) \quad (3.26)$$

For a full description of how the classification factors were modelled you are referred to §3.3.3.

### 3.2.6 Ecotoxicity

Aquatic and terrestrial ecosystems are the end point in the classification of ecotoxicity. Separate classification factors are distinguished for the media water and soil. These result in two effect scores for aquatic and terrestrial ecosystems. These effect scores cannot be added.

For the time being the *exposure factor* for the classification of ecotoxic substances will be assumed to be 1. Hence the classification factor will depend solely on the effect factor. Normally the approach used by the RIVM would be used to calculate the effect factors for aquatic and terrestrial ecosystems. In this approach MTCs (maximum tolerable concentrations) are defined for the media air and soil on the basis of the ecotoxicity of the substances (Slooff, 1992). However, one modification will be made to this. The RIVM uses two methods to calculate MTCs, a *refined method* and a *provisional method*. The

provisional method is an RIVM modification of the *EPA method* (EPA, 1984; Van de Meent *et al.*, 1990). The method used depends on the toxicity data available for each substance. Only the EPA method will be used here as the simultaneous use of two methods is undesirable when defining classification factors given that they will have to be compared with each other. To distinguish them the MTCs calculated in this way will be referred to as "MTC<sub>EPA</sub>". For water the "MTC<sub>EPA</sub>" is expressed in mg·m<sup>-3</sup> water; for soil it is expressed in mg·kg<sup>-1</sup> soil. The *ecotoxicological classification factor for aquatic ecosystems (ECA)* is

$$ECA = \frac{1}{MTC_{EPA}} \quad (3.27)$$

and the *ecotoxicological classification factor for terrestrial ecosystems (ECT)* is

$$ECT = \frac{1}{MTC_{EPA}} \quad (3.28)$$

The method used to calculate these factors for this report is discussed in §3.3.4.

These classification factors can be used in product studies to calculate the effect scores for aquatic and terrestrial ecotoxicity. Based on the emissions  $m_{w,i}$  (in mg) and  $m_{s,i}$  (in mg) to water and the soil the effect score for *aquatic ecotoxicity* (in m<sup>3</sup> water) is

$$\text{aquatic ecotoxicity} = \sum_i ECA_i \times m_{w,i} \quad (3.29)$$

and for *terrestrial ecotoxicity* (in kg soil)

$$\text{terrestrial ecotoxicity} = \sum_i ECT_i \times m_{s,i} \quad (3.30)$$

### 3.2.7 Photochemical oxidant formation

The end point in the classification of this environmental problem is the formation of oxidants in the troposphere. Photochemical oxidant formation, most commonly associated with summer smog, is the result of reactions between nitrogen oxides (NO<sub>x</sub>) and volatile organic compounds (VOC) under the influence of UV light. The more reactive substances will react within a few hours in the vicinity of the emission source (i.e. at regional or national level) while less reactive compounds may spread more before oxidants are formed.

Ozone is considered the most important oxidant. Ozone is often used to represent all oxidants which may be created during photochemical reactions between NO<sub>x</sub> and VOC (RIVM, 1991; Van den Born, 1991). Ozone formation in the troposphere which is discussed here, should not be confused with ozone depletion in the stratosphere. The latter problem was discussed in the section on ozone depletion (§3.2.4).

The contribution made by different types of VOC to ozone production during smog episodes varies greatly. *Photochemical ozone creation potentials* (POCP) were developed to assess the various emission scenarios for volatile organic compounds (Derwent & Jenkin, 1990). A recent UN protocol included the following definition of the POCP (UNECE, 1990): the POCP of an emission is 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. Expressed as a formula:

$$POCP_i = \frac{a_i/b_i}{a_{C_2H_4}/b_{C_2H_4}} \quad (3.31)$$

where  $a_i$  represents the change in ozone concentration due to a change in the emission of VOC  $i$ ,  $b_i$  the integrated emission of VOC  $i$  up to that time, and where the enumerator contains these parameters for ethylene, the reference substance.

The POCP is estimated as the quantity of ozone produced. The quantity of photochemical ozone is determined by means of a model to calculate photochemical ozone production with and without the presence of a given volatile organic compound (UNECE, 1990). This means that, unlike the ODP and GWP potentials, the calculation does not result in an effect due to a *marginal* change in emissions. However, it would appear to be more attractive to calculate POCPs on the basis of a marginal change



in the emissions as well. In the current methods, with or without the presence of a given volatile organic compound, 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  $\text{NO}_x$ . Under the current approach to POCP only VOCs are classified, not  $\text{NO}_x$ . Although this substance is not consumed in the chemical reactions its background concentration affects the course of the ozone production.  $\text{NO}_x$  is also removed from the atmosphere by acid and nitrifying deposition and this substance has to be supplied to maintain photochemical reactions (RIVM, 1992). In principle a POCP could be calculated for  $\text{NO}_x$ . However, the current approach to the calculations, in which the  $\text{NO}_x$  emissions are set to zero would lead to absurd results as there would be no ozone production at all. The emission-effect curve of  $\text{NO}_x$  is far from linear. If POCPs were calculated on the basis of marginal emission changes it would also be possible to derive a POCP for  $\text{NO}_x$ . Finnveden *et al.* (1992) propose the use of subscores: one score for VOC based on the POCP approach and one subscore for  $\text{NO}_x$  based on a kg total of this substance. As mentioned in §3.1.4 we propose to flag  $\text{NO}_x$  emissions to remind us of the photochemical oxidant formation, and to determine the POCP for this substance at which the result of the study would change in the reliability and validity analyses.

An upper and lower value for the POCP for each substance could be used when calculating the effect score for photochemical oxidant formation. The range is fairly wide which indicates the uncertainties currently associated with the POCP approach. This range is partly due to the different methods used to calculate the change in the ozone concentration due to a change in a VOC emission.

POCPs have been determined for many individual volatile organic compounds whose individual emission data are not available, nor are they likely to become available in the near future in an LCA. For most present studies only data about hydrocarbons as a group and about methane will be available. In that case average POCPs will have to be derived from the POCPs of individual volatile organic compounds. POCPs could also be calculated in the same way for common processes such as combustion processes and the processes for the production and use of volatile solvents. However, the average composition of the emissions from these processes has to be known. These average POCPs can be calculated from the arithmetical average of the POCPs for individual volatile organic compounds\*.

The issue of photochemical oxidant formation could be subdivided into two elements:

- smog episodes with peak oxidant concentrations with effects on people, plants and crops;
- a low rise in the average oxidant concentration in the troposphere which largely affects plants and crops.

The POCP approach can be used to quantify the first problem. At present no method has yet been developed to quantify the second problem, hence this element will be excluded for the time being.

We may therefore conclude that when used to quantify part of the problem of photochemical oxidant formation the POCP model requires some major improvements. For the time being we propose using the average POCPs and the range of these values as included in the UNECE protocol, as classification factors. The range in results can be considered as an indication of the uncertainty of the models. The POCP approach will result in one comprehensive score for the emission of different types of VOC:

$$\text{oxidant formation} = \sum_i \text{POCP}_i \times m_i \quad (3.32)$$

$\text{NO}_x$  emissions can also be flagged to ensure they are included in the reliability and validity analyses (see §4.2).

\* According to the UNECE protocol (1990) the POCP could provide a useful indicator for the types of VOC which would have to be dealt with first. It was then decided to quantify the required emission reduction in kg, without further weighting on the basis of POCPs. It is likely that the decision to concentrate the policy initially on reducing the volume of VOC emissions influenced the proposal to aggregate these emissions without weighting, rather than on the basis of POCPs. In classification for an LCA weighted aggregation is needed to take environmental processes and differences in substance properties into account.



### 3.2.8 Acidification

Potential acid deposition\* onto the soil and in water is the end point of the classification for acidification. Potential acid deposition can be expressed as potential H<sup>+</sup> equivalents. Potentially acidifying emissions† of SO<sub>2</sub>, NO<sub>x</sub> and NH<sub>x</sub> can be aggregated on the basis of their potential to form H<sup>+</sup>. In the calculations of H<sup>+</sup> equivalents it is assumed that one mol SO<sub>2</sub> will produce two mols H<sup>+</sup>, that one mol nitrogen oxide compounds (NO<sub>x</sub>) will produce one mol H<sup>+</sup> and that one mol reduced nitrogen compound (NH<sub>x</sub>) will produce one mol H<sup>+</sup> equivalent. Potential H<sup>+</sup> equivalents provide an example of an exposure factor.

As proposed by Vermeire *et al* (1992) an *acidification potential* (AP) can be developed by analogy with the GWP, ODP and POCP. In this way the AP is defined as the ratio between the number of potential H<sup>+</sup> equivalents ( $\nu_i$ ) per mass unit substance  $i$  ( $M_i$ ) and the number of potential H<sup>+</sup> equivalents ( $\nu_{ref}$ ) per mass unit of a reference substance ( $M_{ref}$ ) sulphur dioxide (SO<sub>2</sub>) is proposed as the reference substance. Expressed as a formula‡:

$$AP_i = \frac{\nu_i/M_i}{\nu_{SO_2}/M_{SO_2}} \quad (3.33)$$

Detailed information about the calculation of the AP of each substance is included in §3.3.5. Acidifying substances can be aggregated with the AP using the formula

$$acidification = \sum_i AP_i \times m_i \quad (3.34)$$

This parameter indicates the quantity of an SO<sub>2</sub> emission with the same potential acidifying effect.

### 3.2.9 Nutrification

Nutrification‡ is the addition of mineral nutrients to the soil or water which increase production (Van Straalen & Verkleij, 1991). This definition implies that nutrifying substances could always be a constraint on biomass production. Hence, in principle, any nutrient can have a nutrifying effect when it is added to the soil or water in large quantities<sup>§</sup>.

Nitrogen (N) and phosphorus (P) are the most important nutrifying elements. Carbon (C) and trace elements such as magnesium and silicon are rarely constraints. At present it is not yet clear to

\* For the time being, direct emissions of potential acids into water will not be considered. Such emissions, of many different weak bases and strong acids occur in much smaller quantities and their contribution to the acidification of water is therefore negligible. However, a more complete classification of all potentially acidifying emissions would be desirable in the long term to obtain a comprehensive approach.

† In principle each emission of a weak base may lead to acid deposition. Up to now studies of acidification have been limited to SO<sub>2</sub>, NO<sub>x</sub> and NH<sub>x</sub>, as these are most commonly emitted and the largest causes of acid deposition. Further to this we would like to note that a more complete classification of all potentially acidifying emissions would appear to be desirable in the long term for the sake of completeness.

‡ It might be desirable to include an effect factor in classifications at a lower level of scale. Such an effect factor could be based on the *critical loads* for H<sup>+</sup> which have recently been developed (Hettelingh *et al.*, 1991). The critical load is the quantity of acid deposition which an ecosystem (forest soil, surface water) can tolerate without changes occurring in the chemical composition of the soil, water or tree needles which could lead to ecosystem damage. The critical loads depend on the ecosystem type (e.g. coniferous forest, deciduous forest and surface water) and are based on the latest scientific knowledge in this field. They are comparable with the NOECs for ecotoxicity. When an effect factor is added the potential H<sup>+</sup> equivalents have to be converted to more appropriate equivalents for a given area by using dispersal and deposition models (which should include degradation processes, etc.). These equivalents can then be compared with the critical load for the area concerned. Such models have been developed for Europe (Alcamo *et al.*, 1987) and should also be developed for other Continents. A future spatial differentiation of the acidification classification could be implemented in this way. However, the inclusion of an effect factor would not be useful for the generic classification at global level developed in this study. A global average critical load would not provide any differentiation.

§ Here nutrification is used as synonymous with eutrophication. In this report this includes over-fertilisation of water as well as soil.

¶ The saturation of agricultural land with phosphorus due to excessive use of fertilizer in agriculture, which reduces the fertility of the soil and promotes nitrate leaching to the groundwater, does not fit into this definition of nutrification. Nitrate leaching could be classified under the problem type human toxicity. In LCAs where agricultural activities are important phosphorus saturation of agricultural land could be classified in terms of total phosphorus emissions (kg P).

what extent potassium (K), another nutrient, contributes to nitrification. Hence the classification of nitrifying elements will be limited to N and P compounds.

The effects of nitrification must be distinguished between soil nitrification and nitrification of surface waters. In both media nitrification will result in generally undesirable shifts in the numbers of species in ecosystems and a reduction in the ecological diversity\*. In surface waters the shift in the range of species is generally shown by rapid algal growth. In some seasons when the algae die and at night when the algae consume rather than produce oxygen, this increased algal growth may lead to a lack of oxygen. This can have serious effects on the flora and fauna in the affected body of water.

Another form of oxygen deficit often of a more chronic nature, may be caused by emissions of organic materials which can bind oxygen (a form of biomass emission). These emissions are generally expressed as the *biological oxygen demand* (BOD) or the *chemical oxygen demand* (COD), both of which are expressed in kg O<sub>2</sub>. The oxygen is mainly consumed by the biological degradation of organic matter (C). When assessing nitrification N, P and C (measured as O<sub>2</sub>) can be aggregated by quantifying their *potential* contribution to biomass formation. This aggregation can be based on the average ratio of N and P in biomass and the O<sub>2</sub> required for the breakdown of this biomass. In this way N, P and O<sub>2</sub> are converted e.g. to N equivalents through ratios (the ratios for N, P and O<sub>2</sub> are 7 ÷ 1 ÷ 142). A reference substance could also be used to express nitrification, similar to the ODP, GWP, POCP and AP. In this way a *nitrification potential* (NP) can be derived for all potentially nitrifying substances. Here the NP is defined as the ratio between the potential biomass in N equivalents per emitted quantity of substance *i* (*M<sub>i</sub>*) and the potential biomass in N equivalents per emitted quantity of a reference substance (*M<sub>ref</sub>*) such as PO<sub>4</sub><sup>3-</sup>. Expressed as a formula:

$$NP_i = \frac{v_i/M_i}{v_{PO_4^{3-}}/M_{PO_4^{3-}}} \quad (3.35)$$

Calculation of the NP for a number of nitrifying substances is discussed in §3.3.6. The NP is used to aggregate emissions of potentially nitrifying substances as follows:

$$\text{nitrification} = \sum_i NP_i \times m_i \quad (3.36)$$

Hence, the classification factor for nitrifying substances proposed above is *not* dependent on whatever substance happens to be the constraint in a particular location, as this may vary greatly between locations. Here the classification factor for nitrifying substances is based on the fact that N, P and C all have the potential to contribute to biomass formation. This approach was chosen as the aim is to obtain a general, worldwide classification, independent of local differences and as it is not known which medium (fresh water, salt water, groundwater or soil) an emitted substance will eventually enter.

### 3.2.10 Radiation

Ionising radiation is expressed as the number of atoms disintegrating per time unit, the official unit is the becquerel (Bq). A radiation activity of one Bq corresponds with one disintegration per second. The curie (Ci) was used in the past. The curie was defined as the activity of one gram of radium. Curies can be converted to becquerels as 1 Ci = 3.7 · 10<sup>10</sup> Bq. The radioactivity of a substance is expressed in Bq · kg<sup>-1</sup> or Bq · l<sup>-1</sup>. Radioactivity will always reduce during the course of time. The time over which the radioactivity of a substance reduces by half is known as the half-life of that substance.

Different forms of radiation can be released by radioactive disintegration: alpha, beta, gamma, x-rays and neutron radiation. These forms of radiation may add or remove electrons to or from the atoms they encounter (i.e. ionise them). The extent of the ionisation depends on the type of radiation and the material being irradiated. During the 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 the energy of the radiating material will be fully absorbed by the irradiated material. The energy absorbed

\* The reduction in diversity is due to the fact that increased N and P levels make species which grow rapidly and at an early stage more competitive. Hence these species will displace those which grow slowly and later. This results in a reduction in diversity and the dominance of one or a few species which grow rapidly and early.



per mass unit of material or tissue is the absorbed dose (Jonker *et al.*, 1988).

There are also significant differences between the types of radiation when living tissue is irradiated. For example, a given dose of alpha radiation will produce different effects than the same dose of gamma radiation (Jonker *et al.*, 1988). The difference in harmfulness is expressed in a quality factor. This factor is 1 for beta, gamma and x-ray radiation, 10 for neutron radiation and 20 for alpha radiation. The dose equivalent, expressed in sievert (Sv) is calculated by multiplying the absorbed dose by the quality factor. Hence one dose equivalent of alpha radiation will have the same effect as one dose equivalent of gamma radiation.

In future *ionisation creating potentials* (ICPs) could be developed based on models used to calculate the ratio between the absorbed and equivalent dose of a substance *i* and the absorbed and equivalent dose of a reference substance. However at present this is not feasible as such models are not available, as far as we are aware. As a first step towards this approach a survey could be made of models which might be suitable, followed by the selection of the most suitable model and its modification to calculate ICPS.

In the meantime emissions of radioactive substances could be assessed on the basis of a modified *critical volumes* approach. This would be done by dividing the emission of a radioactive substance by some form of radiation standard for that substance. So far radiation standards have only been defined for workplace exposure (working conditions, see §3.2.14). The *International Commission on Radiological Protection* (ICRP) has set an *annual limit of intake* (ALI) for a large number of substances which might be encountered in the workplace (ICRP, 1979-1982). Given the lack of a better alternative these ALIs could be used for the time being\*. It should be noted that the critical volumes approach is a worst case approach and that differences between substances in terms of half-lives and absorbed doses are not considered.

Classification of this aspect will be impossible as long as emission data on radioactive substances are not included in the inventory analysis. If such data becomes available in future the above suggestions could be used to classify radiation emissions.

### 3.2.11 Waste heat

The end point in the classification of waste heat could be the temperature rise in a city or in water. Waste heat emissions can increase the temperature on a local scale, such as in a city or in a lake. They are unlikely to contribute to global warming on a scale such as that associated with the emission of greenhouse gases. The effects on ecosystems of waste heat emissions into the air are negligible. However, depending on the local conditions, the discharge of waste heat into surface waters could result in a substantial temperature increase and effects on aquatic ecosystems. Our proposal is to include only aquatic emissions of waste heat in the classification, for example cooling water emissions from power stations.

The only feasible way to abstract the local character of this problem is by expressing the waste heat emissions into water in megajoules (MJ). These emissions can be determined by estimating the proportions of waste heat discharged via the cooling water and via the air (based on the energy balance for a plant). As this data is available from the inventory analysis no further action is required in the classification. In essence the classification factor for all waste heat emissions to water is 1:

$$\text{aquatic heat} = E_{\text{water}} \quad (3.37)$$

while it is 0 for all waste heat emissions into the atmosphere.

### 3.2.12 Odour

Odour is a problem when a given concentration of odorous substances is experienced as unpleasant. Whether a smell will be experienced as odour (stench) will depend on the individual exposed to it. However, above a certain emission level all individuals will experience it as an odour. In this section

\* These ICRP documents include extensive lists of ALIs for a wide variety of nuclides. These lists were not included in the guide as this would have required too much time and paper. Also, for the time being it is unlikely that the required emission data for these nuclides will become available in inventory analyses.



the term smell will be used for emissions and the term odour or odour nuisance will be used for effects.

Smell could be defined as the observed difference between a sample of clean air and a sample of contaminated air. The concentration at which it is just impossible to observe a difference is different for each substance, it 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 a given 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. Thus, this is a definition at the smell level not at the odour level. Smell can be measured with some objectivity while individual associations become more important for odour (stench)\*. For this reasons odour threshold - in fact smell threshold values - rather than stench threshold values are determined. This approach will also be followed here.

Odour concentrations and *potential* odour nuisance, i.e. apart from the pleasant or unpleasant associations which may be linked with a particular smell are closely related. However, it is not possible to convert an odour concentration directly into odour nuisance. An odour emission may or may not lead to potential stench or odour nuisance depending on the dispersion of the substance (e.g. deposition and evaporation), chemical conversion (e.g. degradation), stack height, meteorological conditions (e.g. the wind direction) and the distance between an emission source and a residential area.

In the longer term it might be attempted to include some of these processes in the assessment of odour emissions and quantify the contribution of individual substance emissions to odour nuisance relative to a reference substance in a similar fashion as the GWP, ODP, POCP, AP and NP. Gaussian plume models (Anon., 1976) or Mackay models (Mackay, 1991) could be used for the *exposure factor*. Gaussian plume models are particularly suitable for modelling the dispersion of odour emissions through the atmosphere. However emissions of potentially malodorous substances to water could also produce malodorous air after evaporation into the atmosphere. This could be included through the use of Mackay models. The odour threshold values of substances could provide the basis for an *effect factor*. With these two factors the area around an emission source where, theoretically, the odour emissions will just be discernable could be quantified (Hauwert & Keulen, 1990; RIVM, 1991). A more abstract level, independent of local conditions such as the position of a residential area, can be created by defining the affected area in terms of the odour which is theoretically just discernable. In this way a *smell creation potential* (SCP) could be derived for every substance, similar to the potentials for some other substance-related environmental problems. The feasibility and development of this approach will require further investigation.

In the meantime emissions of malodorous substances could be classified using a method similar to the critical volumes approach. This is done by dividing the emission of a potentially malodorous substance by the odour threshold value of that substance. A distinction will have to be made between emissions of potentially malodorous substances to the atmosphere and to water. There are also separate odour threshold values for both media. This is expressed in the following formula:

$$\text{malodorous air} = \sum_i \frac{m_{i, \text{air}}}{OTV_{i, \text{air}}} \quad (3.38)$$

where *malodorous air* is the quantity of air contaminated to the odour threshold value ( $\text{m}^3$ ),  $m_{i, \text{air}}$  the emission of substance  $i$  into the atmosphere (kg) and  $OTV_{i, \text{air}}$  the odour threshold value in air of substance  $i$  ( $\text{kg} \cdot \text{m}^{-3}$ ), and

$$\text{malodorous water} = \sum_i \frac{m_{i, \text{water}}}{OTV_{i, \text{water}}} \quad (3.39)$$

where *malodorous water*  $m_{i, \text{water}}$  and  $OTV_{i, \text{water}}$  represent the same parameters for water. Again, this

\* The following quote from Brasser *et al.* (1985) provides an example of this: "In a study subjects were asked to assess the smell of air with a certain concentration of ethylmercaptan. Most subjects found the smell unpleasant, they associated it with sewers. However, one subject had another view, he associated the smell with onion soup. Both sewer odours and the aroma of onion soup contain mercaptans."

provisional approach is a worst case approach which does not take into account the dispersal and degradation processes, which are largely dependent on the substance concerned\*.

The provisional solution described here is only used in part in the guide. The reason for this is that for most substances there is no uniform odour threshold value. Van Gemert & Nettenbreijer (1977) have made an extensive literature overview of substances and their odour threshold values. The measurements for many substances go back to around 1900 and more than ten different values may have been found. It appears that the measured odour threshold value depends greatly on the questions asked and the measuring method. Van Gemert & Nettenbreijer make a distinction between the detection value and the recognition value. For many measurements it is not clear which of these was measured. The purity of the substance, the rate at which the concentration rises, the number of subjects, etc. are also factors which influence the odour threshold. As a result the values available show wide differences†.

However, for emissions to the atmosphere there is a comprehensive list of odour thresholds determined by a uniform method. This list is used in the provisional method for the classification of potentially malodorous emissions into the atmosphere. Such a list is not yet available for water.

### 3.2.13 Noise

Noise, or noise nuisance, is similar to odour in that a given problem is experienced differently by different individuals. Something considered a nuisance by one individual might be appreciated by another person, loud music, for example. Hence, whether or not sound waves will cause a nuisance depends on the actual situation and the person interviewed. The only way to reach a more abstract level beyond these location and person-specific differences is by aggregating the sound production (in Pa<sup>2</sup>·s) obtained in the inventory analysis (see §2.2) 1÷1 and referring to it as "potential noise nuisance". In this case the classification factor is essentially 1. This ignores the fact that some sound emissions may not cause any nuisance at all (long distance between the source and those exposed) or they may cause a great deal of nuisance in certain environments (e.g. a tunnel for traffic):

$$\text{noise} = G \quad (3.40)$$

where  $G$  is the sound production (in Pa<sup>2</sup>·s). This classification method can be used immediately when the required data is provided by the inventory analysis.

### 3.2.14 Working conditions

A quantitative method for working conditions, in which human health is the end point of the classification, has not yet been developed. In essence a classification of the environmental interventions inside a factory could be developed similar to the classification of environmental interventions outside the factory. Inside the factory we are concerned with human toxicity, radiation, noise, odour and safety. If working conditions were to be included in an LCA these would have to be subdivided into five subcategories, similar to the classification of interventions in the outside environment.

In the indoor environment the exposure pathway to man is much shorter than in the outdoor environment. For this reason the degradation of substances will generally not be considered in the classification of emissions for working conditions. Only exposure by respiration and through the skin are relevant as it may be assumed that workers can be relied upon not to drink solvents or eat chemicals. Exposure through the respiratory system may be assumed to be equal to the emissions to

\* Indirect odour emissions cannot be assessed with the critical volumes approach. Ozone is an example of a potentially odorous substance. Direct ozone emissions are negligible, particularly in comparison with the ozone formed by photochemical reactions of volatile organic compounds and NO<sub>x</sub> (see also the section on the formation of photochemical oxidants). At present these indirect ozone emissions cannot be quantified in absolute terms hence they cannot be included in the classification of odour emissions. It is likely that this problem is similar to the greenhouse effect caused by the indirect formation of CO<sub>2</sub> and O<sub>3</sub> by photochemical reactions of volatile organic compounds and NO<sub>x</sub> (see above). It might be possible to solve this problem if smell creation potentials (SCPs) could be developed to include these reactions, similar to the GWPs.

† Two examples: for H<sub>2</sub>S two determinations from 1924 and 1930 resulted in detection limits of 0.0001 and 0.18 mg·m<sup>-3</sup>; for menthol 0.0004 and 0.9 mg·m<sup>-3</sup>.



the air in the building including, for example, the evaporation of volatile substances from water which may also lead to exposure through the respiratory system. There is no need to correct the emissions for degradation, in contrast with the proposal for the human toxicity exposure factor. Safety measures to protect personnel in the plant would have to be considered when determining emissions and exposure. This will affect the inventory analysis. These details can only be determined for each individual factory and will have to be abstracted in the generic classification. Furthermore an effect factor could be added, similar to the classification of human toxic emissions to the outdoor environment. There are several standards for the maximum daily exposure, such as the list of *maximum accepted concentrations* (MACs; Arbeidsinspectie, 1989) for toxic substances and the list of ALIs for radioactive substance emissions. The use of these lists to provide a basis for the effect factor appears to be more appropriate than for the effect factor for emissions to the outdoor environment, as was proposed above for radioactive substances. Nevertheless the problem remains that these standards are not only based on health considerations but also on economic and technical feasibility. Hence the effect factor for the classification of workplace emissions and the effect factor for similar problems outside the workplace should be as close as possible.

Besides the development of classification factors for working conditions based on the above proposals, this will also require quantitative emission data from the inventory analysis linked to the output produced.

Until these classification factors and the necessary inventory data are available, only qualitative remarks can be made about working conditions. This is the approach taken by the *Technical University of Denmark*\* (Winge, 1991).

We can now draw the following conclusion: the quantitative use of working conditions could be developed in a similar way to that used for problems associated with the outdoor environment, i.e. human toxicity, radiation, noise, odour and safety. Unlike the effect factor the exposure factor would have to be modified in some respects. However, until the inventory analysis can provide the quantitative data required for this approach it will only be possible to make qualitative remarks about each process.

### 3.2.15 Desiccation

Desiccation (or drying out) is a regional problem caused by the abstraction of groundwater and surface water for industrial and potable water supply, agricultural irrigation and water management (control of the water table). Water management would not appear to be associated with the functioning of a product. Desiccation is a problem as it endangers two important functions of water: as an essential human resource and a conditional parameter for nature.

Not all water use contributes to harmful desiccation. The harm associated with its use depends on the local water reserves and the period of time during which the abstracted water cannot be used for other purposes. Water which is only withdrawn temporarily, as cooling water, for salt recovery or for purification, is not harmful in terms of desiccation. However, if the water abstraction level is relatively high compared to the local or regional water reserves these types of water use may be harmful. Hence, the local and regional water reserves have to be considered in the assessment. A distinction can also be made between the use of groundwater and the use of surface water.

To summarize: harmful use of water can be defined as the total water consumption minus non-

\* This method was used in a study of the environmental aspects of a range of materials. However it could also be applied to products. All possible exposure situations (substances, materials) associated with all processes in the life cycle, as well as their state (dust, gas, liquid, vapour, etc.) are listed. The exposure route (inhalation, skin contact, etc.) as well as the extent of the exposure are listed in three categories (high, average, low). The substances are then classified in four groups, depending on their potential toxic effects:

- irritating, corrosive, headache inducing, etc.;
- allergies, bronchitis;
- teratogenic, mutagenic, neurotoxic;
- carcinogenic.

This data provides the basis of the *occupational health profile* which indicates the number of substances in each toxicity category and the level of exposure.



harmful water use; what is non-harmful will depend on the local and regional water reserves. It is also possible to distinguish between the use of surface water and the use of groundwater. Only the total consumption of groundwater and surface water can be quantified in a generic classification at global level. These volumes could be divided by the global reserves of groundwater and surface water, but this cannot be differentiated in the results.

If the inventory analysis provided 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. As was discussed in the section on the depletion of resources, an assessment of water in terms of depletion is not useful as water is not a scarce resource on a global level. Water circulates in a closed system, the main problem with water usage is local or regional desiccation.

For these reasons we propose that water be excluded from in the generic classification at global level. However, desiccation could be included in a classification at a lower level of scale in which data on the groundwater and surface water reserves are available.

### 3.2.16 Physical damage to ecosystems

The data on space use collected in the inventory analysis refers to five ecosystem types. This provides a basis for an assessment of damage to ecosystems due to space use. The different changes in the various ecosystems would have to be weighted in the classification. At present a weighting method of this kind is not available. Such a method could be based on the scarcity of the various categories of ecosystem and the relative value of the ecosystem types to man and nature (life support systems and biodiversity), like the method used for the depletion of resources. The first aspect can be quantified on the basis of the presence of the various types of ecosystems throughout the world. As far as we are aware systematic data about this is not available. It is much more difficult to quantify the second aspect: the relative value of the ecosystem types. Factors for this aspect could, for example, be set by a panel of experts under the auspices of the IUCN. A provisional solution will have to be used while certain factors are unavailable, as it would not be desirable to specify the ten subscores without weighting.

Five categories could be included in two main categories to provide an interim solution: natural (categories I, II and III) and non-natural (IV and V). This division is fairly arbitrary and is no more than a temporary solution\*. This provisional solution can be expressed as:

$$\text{damage} = R_{I \rightarrow IV} + R_{II \rightarrow IV} + R_{III \rightarrow IV} + R_{I \rightarrow V} + R_{II \rightarrow V} + R_{III \rightarrow V} \quad (3.41)$$

where  $R$  represents the quantity (in  $\text{m}^2 \cdot \text{s}$ ) in each category.

### 3.2.17 Damage to the landscape

Landscape damage has not yet been included in an LCA. It is a highly local and aesthetic problem and the perception depends greatly on the observer. It can only be described in terms of a particular element (e.g. a building) which detracts from the view of the landscape at a given location.

In a classification at local level damage to the landscape could be included as a separate problem type although this might be difficult. It would be necessary to specify, quantify and aggregate all elements in the landscape, such as open spaces, ditches, ditch banks, hedgerows, etc.

The IUCN (1982) system of categories, used in the proposed classification of ecosystem damage (§3.2.16) includes an element of landscape assessment. The proposed inclusion of damage to ecosystems and damage to the landscape have now been integrated in a single step.

Damage to the landscape will not be included separately in the classification while there is no acceptable method for this. Our proposal for the generic classification at global level is to develop this issue together with damage to ecosystems based on the five land-use categories referred to above.

\* The definition of four unit changes: I→II, II→III, III→IV and IV→V is another option. A change of 1  $\text{m}^2 \cdot \text{s}$  from category II to IV would then be included as 1  $\text{m}^2 \cdot \text{s}$  II→III and 1  $\text{m}^2 \cdot \text{s}$  III→IV. See also §2.2.2.

### 3.2.18 Direct victims

Victim categories would have to be weighted in the classification if other victims besides fatalities (e.g. slightly injured, seriously injured) were included in the inventory analysis. As far as we are aware there are no methods for this. Such a method will not be necessary while data about slightly and seriously injured victims is not available. For this reason the inventory analysis data is included in the classification with an effect factor 1:

$$victims = S \quad (3.42)$$

where  $S$  is the number of victims (dimensionless) listed in the inventory table.

### 3.2.19 Qualitative environmental effects

The qualitative environmental interventions included in the inventory analysis can also be included in the environmental profile. They could be accompanied by a discussion about the estimated or expected effects of the interventions. Often there will be data about substance emissions which are known to contribute to certain problems but for which classification factors are not yet available. Examples include substances which make an indirect contribution to the greenhouse effect ( $CH_4$ ,  $CO$ ,  $NO_x$  and  $NMHC$ ), substances which may deplete the ozone layer, but for which no ODP has been defined ( $N_2O$ ,  $NO$ ,  $CH_4$  and  $CO_2$ ) and  $NO_x$  as a contributor to the formation of photochemical oxidants.

## 3.3 Development of the classification factors

A provisional guideline for the classification of environmental interventions by their potential effects has been developed in the form of a set of tables with classification factors, as well as the formulas to be used with these tables. These tables and formulas are included in the guide. Their justification, in scientific terms, is included in §3.2. Six of these effect scores are interesting in their own right and are not included in either of the parts. These are discussed below.

### 3.3.1 Depletion of resources

The depletion of resources is at the heart of the discussion about sustainability and is a problem in its own right. Depletion may lead to a loss of functions and potential functions of certain material applications. Furthermore the continued use of resources may lead to a shift to poorer or less favourably sited reserves thus resulting in greater emissions, damage, desiccation, etc. For these reasons it is useful to include the aspect of depletion in the classification of an LCA.

Depletion depends on limited reserves of resources as well as the extent to which these resources are used. If the reserves of a resource decline, or if consumption exceeds its replacement rate then it may be depleted. Here we will limit ourselves to resources whose depletion will become, or still be, a problem within the next 100 years. Lists of the relevant resources are included in the guide. These lists are provisional. Some resources are still being considered for addition to the list. Other resources may be removed from the list in future, either because the application is disappearing because they have been replaced, or because new reserves have been found or can be recovered, or because the resource is actually depleted.

A distinction has to be made between the utilization of resources which contributes to depletion (*consumption* of resources) and that which does not contribute to depletion (*use* of resources). The consumption of resources is largely associated with applications in which waste is disposed of, the quality deteriorates with recycling and useful application, diffuse dispersal or dissipative use. By contrast the use of resources includes the utilization of cooling water, wind energy, etc. The way in which reuse, including any quality deterioration, is accounted for is essential. This topic will require further consideration.

Another important distinction is that between *biotic* and *abiotic* resources. Biotic resources which are grown and harvested on a sustainable basis should be excluded. By themselves they do not contribute to depletion. Any other substances required for this process, which are relevant in the context of depletion, should be included in an LCA.

This issue has two main dimensions: the utilization of resources which may be depleted, and the



environmental consequences of the continuing utilization of resources. At present the environmental consequences of the continuing utilization of resources can only be included in the sensitivity analysis of an LCA.

One of the problems with putting a value on the utilization of resources is that at present there are no definite standards which can be used by contrast with toxicity, for example. There are several options for weighting the use of resources. The following options could be considered:

- comparing the consumption of a resource with its reserves;
- comparing the use of resources with the annual production of that resource;
- comparing the use of a resource with the reserves/production ratio of that resource;
- comparing the use of a resource with both the reserves and the reserves/production ratio of that resource.

All these options will be developed below. The first three methods all have certain advantages and disadvantages. The fourth method was developed to remove these disadvantages.

#### *Weighting based on the reserves (1)*

Three types of reserves are relevant when comparing the consumption of a resource with its reserves: the overall global reserves, the total available recoverable reserves and the recoverable reserves. The weighting method is the same for all three:

$$\text{depletion} = \frac{\text{consumption}}{\text{reserves}} \quad (3.43)$$

thus the extent of depletion is expressed as a dimensionless parameter.

When the first option is used the relative significance of the consumption of resources can be indicated by comparing the consumption to the global reserves of that resource. The total global reserves consist of the reserves not yet recovered, reserves currently in use and reserves which have been disposed of. In practical terms a major disadvantage of this option is that no reliable and meaningful data are available on the total global reserves. There is no data about the reserves which are currently in use and those disposed of, while the data about unrecovered reserves is inadequate or unavailable.

The second option is to compare the utilization of resources to the total available and recoverable reserves. The advantage of such an approach is that the actual problem is considered, i.e. if and for how long a resource is or will be available. Again, the disadvantage is that there is no data about the total global reserves which are recoverable and available. Information about recoverable reserves is normally available. There is no data available about the reserves which are in use or have been disposed of. Another disadvantage of this option is that recoverability and availability depend on economic and technical developments.

The third option is to compare the utilization of resources only with the recoverable resources. A major practical disadvantage of this is that the data about recoverable reserves are based on the economic and technical recoverability. This makes it sensitive to changes in prices. Another disadvantage of this approach is that not all available reserves are considered and that the problem is therefore only partly covered. However, the advantage of this option is that there is data available about the recoverable reserves. The *probable reserves* provide the best data as these relate to the reserves which are or will be recoverable within a period of thirty years.

At present it is not possible to include the total world reserves in resource utilization. Only the third option, comparison with the recoverable reserves, is practical. However, such an approach does not address the actual problem, i.e. how long a given resource will continue to be available\*. This is an important methodological disadvantage. A further disadvantage is that recoverable reserves can be influenced by technical and economic developments.

\* The extent of over-exploitation will have to be included in the weighting, particularly for biotic resources.



*Weighting based on the annual production (2)*

The relative significance of the utilization of resources can also be indicated by comparing it with the actual changes in the reserves. This can be expressed by a formula:

$$\text{depletion} = \frac{\text{consumption}}{\text{annual production}} \quad (3.44)$$

In this way depletion is expressed in yr. An argument in favour of this weighting method is that the annual production is a measure of the availability of the resource. As a result the depletion period is the same for all resources. This flaw could be remedied by classifying the resources in categories (e.g. scarce, somewhat scarce, not scarce) based on their depletion period. Aggregation within the categories might be useful. The various categories could remain unweighted to provide a set of subscores or a weighting factor could be applied to them.

The actual change in reserves is the total world consumption. This method requires data about the total consumption of resources (disposal after reuse). Such data is not available. Alternatively world use could be considered instead of world consumption. The actual world use could be assumed to be equal to the total recovery and production of resources. Further investigation would be needed to demonstrate to what extent world production is a good indicator of the utilization of resources. A disadvantage of this is that data about the actual world consumption is not available and that the world use would still have to be used. The advantages are that the proportion is weighted, it is compared with the decrease in reserves and that there is reliable information available about the world production.

A disadvantage of this method, in methodological terms, is that there is an arbitrary allocation to categories which can only be aggregated in a fairly subjective manner.

*Weighting based on the reserves/production ratio (3)*

In the third method the use of resources is compared to the reserves/production ratio (the *r/p*-ratio). This is a measure of the sufficiency of the reserves: the number of years over which the reserves will suffice at the current production level. This is calculated as follows:

$$\text{depletion} = \frac{\text{consumption}}{\text{reserves-production ratio}} = \frac{\text{annual production}}{\text{reserves}} \times \text{consumption} \quad (3.45)$$

to create an effect score with the dimension  $\text{kg} \cdot \text{yr}^{-1}$ .

A major advantage of this method is that the consumption of resources is linked directly to scarcity. However, the share in the consumption of resources is not weighted which may lead to strange conclusions\*. Another disadvantage is that the nature of the unit in the effect score is chosen fairly arbitrarily. It would be equally valid to choose the volume, as a result of which resources would be weighted in different ratios.

*Weighting based on the reserves and the reserves/production ratio (4)*

All the options available for weighting the utilization of resources have methodological disadvantages. The disadvantage of weighting based on the reserves is that the depletion period is not taken into account. However, the disadvantage of using the reserves/production ratio is that the extent of the reserves is disregarded. Both methods could be combined to benefit from their advantages. This is done as follows:

$$\text{depletion} = \frac{\text{consumption}}{\text{reserves} \times \text{reserves/production ratio}} = \frac{\text{annual production}}{\text{reserves}^2} \times \text{consumption} \quad (3.46)$$

resulting in an effect score with the dimension  $\text{yr}^{-1}$ .

This solves the methodological disadvantages of the three methods discussed above. However, the

\* For example: let us assume that the *r/p* ratio of resource A is 30 years and that of resource B 100 years. In an LCA the use of 1 kg of resource A would be less important than the use of 10 kg of resource B when the *r/p* ratio is used as an indication of the use of resources. This would still occur even if the use of 1 kg of A amounted to the entire available world reserves and the use of 10 kg of resource B amounted to a minuscule proportion of its reserves.

new method has a major practical disadvantage: it requires data about the reserves as well as the annual production. As the reserves are squared any inaccuracies will have a large effect.

**Implementation**

Below we will select the recoverable reserves and reserves/production ratio (i.e. method 4) as the best approach in methodological terms. This is clearly the best approach for biotic resources as both the utilization of the resources as the extent of global over-exploitation are decisive factors.

Because of the practical problems different methods will be used for abiotic and biotic resources. These problems are due to the fact that the reserves (of which there are only very rough and debatable estimates) are squared in the weighting process. Hence for abiotic resources we will limit ourselves to weighting on the basis of the recoverable reserves and not consider the extent of exploitation or over-exploitation.

The *World Resources Institute* (WRI) provides data about the recoverable reserves: the *world base reserve* (WBR)\*. These data concern the *measured* and *indicated* reserves, which will be economic, marginal or uneconomic to mine over the next thirty years. Figure 3.4 illustrates the way in which reserves are classified. The shaded area represents the reserves used here.

	proven reserves			unproven reserves	
	extent of reserves		possible	hypothetical	speculative
	certain	likely			
economically recoverable	reserves				
not economically recoverable	resources				

FIGURE 3.4. Overview of the types of reserves. The shaded area "reserves" is used in the definition of the classification factor.

Use of the WBR also has some problems. Data is not available on all resources. In some cases the data file was supplemented from other sources when WBR data was not available from the WRI.

Both the reserves and the rate of depletion (the nett reserves/production ratio) can be determined for certain biotic resources. A standard factor can be used if the data to determine the *r/p* ratio is not available.

**Discussion**

It might be worth considering the use of a single parameter to classify the depletion of biotic and abiotic resources combined. Furthermore, it would appear to be a better form of classification to compare the available quantity of resources with the total global reserves which are available and recoverable.

As indicated above the data about global reserves is too uncertain. This is also illustrated by Figure 3.5. Many future developments, in particular, could have major effects.

A comparison with the average global annual production rather than with the reserves could be considered as the data on the reserves is highly uncertain and variable.

The global annual production determines the rate at which resources are depleted. The share in the depletion rate can be weighted by calculating a resource score on the basis of the global annual production. This would provide an indication of the contribution to the depletion. A further advantage

\* The *base reserves* are the mineral reserves of such concentration, quality, thickness and depth that they are recoverable with current mining and production techniques. This includes both *measured* and *indicated* reserves and covers reserves which are economic, marginal or uneconomic.

source	geological availability			
	known			undiscovered
	certain	proven	likely	
conventional	<ul style="list-style-type: none"> <li>● reserves</li> <li>● stocks</li> <li>● scrap</li> <li>● etc.</li> </ul>			hypothetical      speculative      economic
unconventional	<ul style="list-style-type: none"> <li>● potential (unrecycled) byproducts</li> <li>● erosion products</li> <li>● oceans</li> <li>● household and industrial waste</li> <li>● mine tailings, etc.</li> <li>● substitute materials</li> <li>● rock</li> <li>● extraterrestrial materials</li> <li>● undiscovered materials</li> </ul>			uneconomic

FIGURE 3.5. Overview of the sources of resources.

would be that this would make it possible to combine biotic and abiotic resources in a single parameter.

Further depletion will be associated with environmental consequences and risks particularly when poorer reserves, reserves with other contamination and from other locations involving more transport have to be used. In the long term this will lead to other environmental interventions (energy consumption, emissions, space use, desiccation, damage, etc.). Another quantified product comparison could be made for these environmental consequences and risks, based on the most sensitive resources. This should include options for substitution, prevention, reuse and the exploitation of other reserves. In the current LCA method this could be done in the sensitivity analysis\*. It will be necessary to consider whether this should be developed as a separate indicator for the aspect of resources, as it is one of the most marked consequences of the further utilization of resources.

### 3.3.2 Toxicity: the intended model and the provisional model

To classify emissions by toxicity the ratio between the classification factors of the substances concerned has to be an accurate reflection of the ratio in the toxicity of these substances. Existing standards cannot be adopted directly as other factors besides toxicity were often considered. Examples of factors relevant to such standards which should be excluded when classifying substances by their individual toxicity include nuisance odours and loss of flavour in fish due to the intake of certain compounds and natural background concentrations. To determine the correct ratios between the values they should also have been obtained in a similar manner. For this reason the use of different methods depending on the information available has been avoided even though this approach is not uncommon in the policy on standards.

Classification by toxicity distinguishes between human toxicity (§3.3.3) and ecotoxicity (§3.3.4). Ecotoxicity distinguishes between toxicity to aquatic and terrestrial ecosystems.

The critical volumes method which has been used for the classification of toxic substances in many

\* In addition to the changed resource situation a scenario based on future technology would have to be considered. Not only will there be more contaminated fuel sources but there will also be cleaner combustion technology. This would be an interesting exercise, particularly when discussing the future. It could be used as a factor to weight the importance of the depletion of resources. However at present this is not feasible in a normal product comparison.



studies, uses effect factors which are not based exclusively on toxicity. The human toxicity concept did not include a link between the exposure route (air, drinking water or soil) and processes in the environment (degradation, transport) and the actual or potential exposure. As a result the effect scores for human toxicity due to the pollution of air, water and possibly soil could not be added to provide a single score for human toxicity. Even individually they did not provide an accurate representation of the actual situation. CML has developed a new model to meet these objections. This will only be discussed here in general terms as this model could not be set up in the short term. For further information you are referred to Guinée & Heijungs (1992). A simplified version of the model was implemented and will be developed in §3.3.3 and §3.3.4.

In the intended model for the classification of toxic emissions the HTP (*human toxicity potential*), AETP (*aquatic ecotoxicity potential*) and TETP (*terrestrial ecotoxicity potential*) are calculated in a method similar to that used for the ODP and GWP. HTP, AETP and TETP are indices of the human toxic or ecotoxic effect of an emission of a substance relative to an equal emission of a reference substance. Like the ODP and the GWP the HTP, AETP and TETP are relative, dimensionless parameters. The HTP, AETP and TETP are each the product of two factors: the *exposure factor* and the *effect factor* (see Figure 3.2 on page 65). Thus the extent of potential exposure to a substance due to an emission to a given environmental medium and the extent of toxicity of that substance in the event of direct or indirect exposure in that medium are distinguished.

The inclusion of environmental processes and exposure routes in LCAs was first proposed by Vollebregt (1992) in a study of the environmental effects of washing and cleaning agents. Although her proposals were not included in her reports due to some of their disadvantages, her work was very important in the development of the basic concept for the model described here.

The exposure factor represents the exposure level which an emission of a given quantity of a substance would produce relative to the exposure level caused by the emission of an equal quantity of a reference substance. The exposure factor depends on the environmental medium in which the emission occurred, the processes which occur in the environment before the substance reaches the ecosystem or man. Human exposure also depends on the routes by which the emitted substance reaches man, either directly or indirectly. We will use Mackay's (1991) *multi-media environmental models* to model the processes occurring in the environment after emission. A complication arises as these models are based on *emission fluxes* (in  $\text{kg}\cdot\text{yr}^{-1}$ ), whereas in an LCA only *emission pulses* (in kg) are determined. However, it can be proven that this problem can be solved by the introduction of a reference substance (Guinée & Heijungs, 1992). The resulting exposure factor of a substance represents the ratio between the potential exposure level which would be caused by a standard emission of that substance and the potential exposure level associated with an equal emission of the reference substance.

The effect factor of a substance forms the most accurate possible indication of the toxicity of that substance. Naturally it is determined relative to the same reference substance as the exposure factor. The effect factor of a substance specifies the ratio between the potential toxic effect of a standard quantity of that substance and the potential toxic effect of an equal quantity of the reference substance. The effect factors are calculated using both extrapolated data from toxicological animal experiments and empirical data about the effects of human exposure.

When this report was published it was not possible to determine the exposure factors for a number of substances using the method described above. This is because some data about the fate of a number of substances in the environment have yet to be compiled and processed. Hence *provisional exposure factors* have been used here. The method can already be used to determine the *effect factors* of a large number of substances. However, as reference substances are not used with the provisional exposure factors, the associated effect factors should also be specified without the use of a reference substance. Hence *provisional effect factors* will also be used. The provisional effect factors differ from the final effect factors by a factor which is the same for all substances, i.e. their ratios remain the same. *Provisional classification factors* for human toxicity and ecotoxicity have been calculated on the basis of the provisional exposure factors and the provisional effect factors.

### 3.3.3 Human toxicity

#### *Principle: the provisional exposure factor*

The calculation of exposure factors requires data about:

- the size of the parts of the environmental media air, surface water and soil in which the emissions disperse in each of those media;
- the way in which emitted substances are distributed across the media due to exchange between the media (partition);
- the average degradation or immobilization rate of the emitted substances in each environmental medium;
- the routes by which man potentially takes up the emitted substances (directly or indirectly) and the daily intake from each medium;
- the number of people who are potentially exposed every day to the emissions being studied.

The larger the part of the medium in which an emitted substance is dispersed the more it will be diluted and the lower the exposure will be. To approximate the size of the parts of the media air, water and soil in which the emissions will disperse we will use the model developed by Mackay. This model is based on a *unit world* with an area of one square kilometre (Mackay, 1991). This *unit world* can be expanded into a *model world* by multiplying it by the surface of the earth. The model world is an approximation of the size of the environmental media in which substances are dispersed and exchanged. It is assumed that a substance present in one of these components will disperse homogeneously in this medium throughout the world. The Earth's surface area is assumed to be  $5 \cdot 10^8$  km<sup>2</sup> (Harte, 1988).

The part of the atmosphere in direct contact with the earth is known as the troposphere. Its thickness is about 10 km. In the calculations we will assume that there is a uniform pressure of 1 atmosphere. As a result the troposphere in the model world will be compressed to a thickness of 6 km. Hence the total volume of air in the model world is  $3 \cdot 10^{18}$  m<sup>3</sup>.

Approximately 70% of the earth's surface is covered with water. It is assumed that only the upper 10 m will be accessible to pollution in a relatively short term period. Hence the model world contains a total of  $3.5 \cdot 10^{18}$  l water.

Approximately 30% of the Earth's surface is covered by soil. Assuming that pollution will largely be limited to the top 15 cm of soil and that the average soil density is 1200 kg dry matter per m<sup>3</sup> (Mackay, 1991) the model world contains  $2.7 \cdot 10^{16}$  kg soil (dry matter).

A substance emitted into a given environmental medium will generally also enter the other media through processes such as deposition, evaporation, solution and sorption. This process is known as *partition*. Due to partition the concentration of an emitted substance in the medium into which the original emission took place is reduced, while the concentration of that substance in the other media increases. At present partition cannot yet be incorporated in the exposure factor for emissions into the atmosphere and water. Partition of soil emissions has been included in the provisional model although not all transport routes have been included.

The higher the rate at which an emitted substance is degraded or immobilized in the environment the lower the potential exposure will be. After an emission persistent substances such as heavy metals and DDT will remain in the environment essentially forever. They may still pose a threat to human health long after the emissions have ceased. However, toxic emissions which are degraded in the environment before they reach man do not threaten human health despite their toxicity. At present degradation and immobilization have not yet been incorporated in the provisional exposure factor\*.

\* The concentration of an emitted substance in the various environmental media depends not only on the scale of that emission and the size of the medium, but also on the rate at which that substance enters the medium (by direct emission or partition) and the rate at which it disappears (by partition, degradation or immobilization). Given a constant emission flux of a degradable substance an equilibrium distribution will be created among the media. The equilibrium situation will depend not only on the physical chemical properties of the substance but also on the flux and the degradation and immobilization rates. An emission pulse will lead to a temporary rise in the concentration in one or more media. In this event the concentrations in the media will be time-dependent. Hence the exposure factor cannot be calculated simply on the basis of the concentration. The ratio between the exposure factor due to an emission pulse of a given substance and the exposure



In essence this means that all substances are considered as persistent.

With regard to the potential exposure routes for the media air and water only direct intake through the medium concerned has been modelled so far. It is assumed that a person breathes 20 m<sup>3</sup> of air per day and consumes 2 litres of water (Vermeire, 1991). This is based on the consumption of untreated surface water\*. Any air purification (e.g. by air conditioning) will also be disregarded.

A more complex approach is used for potential exposure to soil contamination as this is generally taken up by man indirectly (e.g. by eating crops which contain pollution from the soil). The extent to which man is indirectly exposed to soil pollutants depends on the substance because the way in which substances are transported in the environment depends on the properties of that substance. For example, some substances will be taken up by plants in relatively large quantities while another substance might enter supposedly purified drinking water by permeating plastic water pipes. Van den Berg (1991) has modelled and quantified a large number of routes which contribute to man's exposure to soil pollution. In the model it is assumed that substances in the soil are affected by the following transport mechanisms: transport to arable crops through take up by the plants and the deposition of contaminated soil particles on plants, transport to the atmosphere through transpiration and wind-blown soil particles onto which these substances are adsorbed and migration to purified drinking water by permeation through plastic water pipes. The exposure routes considered for direct and indirect exposure are: ingestion of soil and dust, dermal contact with soil and dust, consumption of crops, inhalation of air, inhalation of soil particles, consumption of drinking water, dermal contact with drinking water in the bath or shower and vapour inhalation in the bath or shower. The provisional *human toxicological C values* for soil based on the TDI have been calculated with the model for the substances under consideration (Van den Berg, 1991).

The number of people exposed every day to emissions via the environmental media also affects the potential harmful effects of those emissions on health: as the Earth's population increases a harmful emission will have more actual or potential victims. In this model the global population is assumed to be 5000 million.

For the media air and water the provisional exposure factor for each medium is the same for all substances. This factor provides a correlation between the quantity of an emission to a medium and the extent to which the world population is potentially exposed to the emitted substance. A different method is used for soil because indirect exposure which depends on the substance also occurs. Hence the exposure factor for soil will be different for each substance.

#### *Principles: the provisional effect factor*

The harmfulness of a given substance to human health depends not only on the toxicity of that substance and the quality of the substance man is exposed to, but also on the exposure route: oral (food and water), inhalation (respiratory tract) or dermal (skin contact). For example, whether there is a risk of lung cancer due to exposure to asbestos depends on whether the exposure was by inhalation or orally. The differences in harmfulness between oral and inhalation exposure are known for only a relatively small number of substances (Vermeire *et al.*, 1991). This data relates either to a difference in the biological availability of the substance concerned (oral exposure may introduce the substance more effectively in the circulation than exposure by inhalation, or vice versa) or on effects which depend on the entry route (the substance's effects after exposure by inhalation differ from those after

factor of an equally large emission pulse of a reference substance can be determined by assuming that the ratio between the exposure factors of two different substances is not affected by the difference between emission fluxes and pulses. Thus the exposure factor of a substance can be expressed as an exposure equivalence relative to the reference substance. This approach lies at the heart of the intended model for the classification of substances toxic to humans (Guinée & Heijungs, 1992).

\* The reason for this is that the fact that surface water is generally treated before human consumption should never justify the discharge of toxic substances into the surface water. In essence this is an extension of the principle of multi-functionality for soil (Kamerstukken II, 1985) to the medium water: in principle all surface water should be clean enough to be fit for human consumption. Alternatively emissions into water could be assumed not to have any toxic effect on humans as these impurities will be removed before consumption. By definition this would make human toxic emissions into water less serious than similar emissions into the atmosphere as they would be assumed not to have any human toxic effects.



oral exposure). Generally dermal exposure is of minor importance and will only be included in the provisional exposure factor for soil.

The ADI (acceptable daily intake) values set by the WHO and the TDI (tolerable daily intake) values set by the RIVM by the same method have been used as a measure of the harmfulness of substances after oral exposure (WHO, 1987; Vermeire *et al.*, 1991). The ADI or TDI is the quantity of a substance which a person can ingest orally for life on a daily basis without a significant risk of harmful effects on health. It is considered the maximum daily intake.

The RIVM has developed TCL (acceptable concentration in air) values for a relatively small number of substances (Vermeire *et al.*, 1991). The health effects of continuous daily exposure by inhalation to air polluted to the level of the TCL are assumed to be equivalent to a daily oral dose of the TDI of that substance.

Generally TDI and TCL values (and ADI values) are determined by extrapolation of data from animal experiments. Such data (e.g. NOECs or LC<sub>50</sub>s) are divided by one or more uncertainty factors, depending on the volume and quality of the data available. The following are common uncertainty factors (Vermeire *et al.*, 1991):

- a factor 10 for the uncertainty due to extrapolation from laboratory animals to man (the *interspecies factor*);
- a factor 10 for the uncertainty due to differences in sensitivity among the population (the *intraspecies factor*).

Other uncertainty factors are also applied as a result of which the resulting *safety factor* may be as much as 1000 or more. Vermeire *et al.* stress that the safety factor provides a reasonable estimate of the *maximum* uncertainty interval, but that the uncertainty factors used are generally too high an estimate of the actual differences in sensitivity as a large safety margin is applied when the estimates are made. Reference is made to Dourson & Stara (1983) who argue, on the basis of animal experiments with a range of substances, that a good estimate of the interspecies factor between, for example, rats and dogs should be 5 to 6 based on the actual average difference in sensitivity (i.e. excluding the safety margin) and between dogs, apes or rabbits and man 2 to 3. On toxicological grounds they also calculate an intraspecies factor of 2 to 3.

Assuming that the uncertainty factors used are generally higher than could be justified on purely toxicological grounds the chance that the safety factor is too high (in toxicological terms) increases as the number of uncertainty factors used to calculate the safety factor increases. Hence that chance will also be higher as the safety factor increases (this is because the safety margin of each uncertainty factor is also included in the multiplications).

As TDI and TCL values include safety margins which are different for various substances the ratio between various TDI and TCL values gives a somewhat skewed representation of the actual toxicity ratios of the substances concerned. Therefore in 1991 Vermeire *et al.* proposed correcting the TDIs and ACAs for this error by multiplying them by a correction factor *N* (the *uncertainty reduction factor*). *N* would be 1, 2, 3 or 4. The value of the uncertainty reduction factor was dependent on the value of the safety factor. Van den Berg (1991) developed this proposal for different substances and used it to draw up provisional human toxicological C values. However, this concept was later rejected. Discussions with Vermeire showed that given current understanding the ratios between the *uncorrected* TDIs provide the best possible reflection of the toxicity ratios of substances. Although the final human toxicological C values had not been released by the time this report was published the most recent advances have been anticipated by reversing uncertainty reduction factors in the human toxicological C values published in 1991: each C value was divided by the relevant uncertainty reduction factor *N*.

No ADI, TDI or TCL has been defined some substances as they do not have a human toxicity threshold: any dose or concentration is potentially toxic. Genotoxic carcinogenic substances are a typical example. In the brochure "Omgaan met Risico's" (VROM, 1988c) the dose or concentration which results in the risk of one additional cancer per 10,000 individuals exposed is defined as the *maximum acceptable risk level*. In this document it will be considered as equivalent to the TDI or the

\* At a later stage the inaccuracy due to the use of excessive safety factors and unequal safety margins will have to be considered in greater detail.

TCL, in accordance with Vermeire *et al.* (1991). This value will be referred to as the TDI or the TCL.

The TCL values will be used to calculate the provisional effect factor for exposure by inhalation (Van den Berg, 1991). For substances for which no TCL value has yet been defined but for which the WHO has set an *air quality guideline* (WHO, 1987) that guideline will be considered as equivalent to the TCL. The acceptable daily intake via air is calculated from the TCL or the air quality guideline. In essence this means that a form of TDI for intake via the atmosphere is calculated. If there is neither an TCL value nor an air quality guideline for a substance the ADI or TDI will be used to determine the provisional effect factor. As in the RIVM method it is assumed that the effect of a substance is independent of the exposure route (oral or by inhalation), that the daily respiratory volume is 20 m<sup>3</sup>, that the bio-availability is the same in both routes and that the average human body weight is 70 kg.

The ADI or TDI is used for the oral intake of pollutants through drinking water. Where both an ADI and a TDI value were available for a substance and they differed, the TDI was used because the TDI values provide the basis for the human toxicological C values for soil. This makes it possible to implement a provisional classification factor for the soil using the TDI, which is not possible with the ADI (see page 96).

#### *Principles: the provisional classification factor*

As neither partition between environmental media nor degradation and immobilization in the environment have been included in the model yet, the concentration of an emitted substance in a given environmental medium is at present assumed to depend only on the absolute magnitude of the emission (emission *pulse*) and not on the rate at which that substance disappears from the medium (due to partition, degradation and immobilization)\*. This makes it possible to use a simplified approach for the calculation of the provisional classification factor which does not necessitate a comparison between the toxic effect of a substance and the toxic effect of a reference substance. The mathematical aspects of the procedure for the calculation of the provisional classification factors of emissions into the atmosphere, water and soil will be developed below. When an LCA is carried out with the aid of the provisional classification factors the scores for human toxicity due to emissions into the media air, water and soil can be added to form a single general score for human toxicity.

#### *Development: provisional exposure factor for air*

The first step in calculating the provisional exposure factor for emissions into the atmosphere is to calculate the link between the magnitude of an emission into the atmosphere of any substance and the resulting concentration of that substance in the atmosphere. The following presumptions are made in the provisional method:

- a substance emitted into the atmosphere will disperse homogeneously through the entire medium air in the model world;
- the transport phenomena of a substance released into the atmosphere to water and soil are not considered;
- degradation and immobilization processes are not considered.

Hence the atmospheric concentration  $C_{in\ a}$  (in kg substance·m<sup>-3</sup>) of a substance due to an emission into the atmosphere  $m_a$  (in kg) of that substance will be

$$C_{in\ a} = \frac{m_a}{V_a} \quad (3.47)$$

where  $V_a$  (in m<sup>3</sup>) represents the total volume of air in the model world.

The *daily dose*  $\dot{m}_{in\ a}$  (in kg substance·day<sup>-1</sup>·person<sup>-1</sup>) of an emitted substance which a person is exposed to on average as a result of the emission concerned depends on:

- the above concentration in air  $C_{in\ a}$  of the substance as a result of that emission;
- the volume of air  $\dot{V}_a$  (in m<sup>3</sup>·day<sup>-1</sup>·person<sup>-1</sup>) inhaled on average per person per day.

The average daily dose per person is

\* Hence the concentration of a substance in a medium is assumed to be independent of time.

$$\dot{m}_{\text{via a}} = C_{\text{in a}} \times \dot{V}_a = \frac{m_a \times \dot{V}_a}{V_a} \quad (3.48)$$

The *total daily human exposure* due to a given emission into the atmosphere depends not only on the daily dose per person but also on the number of people exposed. As it is assumed that every emission will disperse uniformly throughout the total global air volume the following assumption is made:

- each emission will lead to a uniform exposure of the entire world population  $W$  (in persons). Thus the total human exposure per day  $T_{\text{via a}}$  (in kg substance·day<sup>-1</sup>) due to a given emission into the atmosphere is

$$T_{\text{via a}} = \dot{m}_{\text{via a}} \times W = \frac{m_a \times \dot{V}_a \times W}{V_a} \quad (3.49)$$

This can be interpreted as the total quantity of the substance to which the entire world population would be exposed per day as a result of the emission.

The *provisional exposure factor for air*  $X_a$  is the factor by which a given emission into the atmosphere  $m_a$  has to be multiplied to obtain the total daily human exposure  $T_{\text{via a}}$ . This is obtained by dividing  $T_{\text{via a}}$  by the emission  $m_a$ . In view of the above it can be expressed as

$$X_a = \frac{T_{\text{via a}}}{m_a} = \frac{\dot{V}_a \times W}{V_a} \quad (3.50)$$

where:

- $X_a$  = provisional exposure factor air (day<sup>-1</sup>);
- $\dot{V}_a$  = volume inhaled by man (= 20 m<sup>3</sup> air·day<sup>-1</sup>·person<sup>-1</sup>);
- $W$  = world population (= 5·10<sup>9</sup> persons);
- $V_a$  = volume of air in the model world (= 3·10<sup>18</sup> m<sup>3</sup>).

Given the current assumptions the provisional exposure factor for air is 3.3·10<sup>-8</sup> day<sup>-1</sup>. This can be interpreted as the fraction of an atmospheric emission which the entire world population would be exposed to per day as a result of that emission.

#### *Development: provisional effect factor for air*

The TCL values developed by the RIVM can be converted to values which represent the *acceptable daily intake via the air*, by using information about the average daily inhaled volume  $\dot{V}_a$  and the average human body weight  $M$ . The values obtained in this way can be considered as TDI values for intake by inhalation. Here they will be indicated as  $TDI_a$ . The acceptable daily intake by inhalation of a substance is inversely proportional to the magnitude of the toxic effect after inhalation of that substance. Hence the inverse of this value corresponds with the toxic effect, and this will be used as the provisional effect factor  $E_a$ .

If there is no TCL for a substance but an *air quality guideline* (AQG) has been defined by the WHO then this value will be used in the same way as an TCL. TCL values and air quality guidelines based on odour thresholds will not be used. The relationship between TCL and  $TDI_a$  can be expressed as

$$TDI_a = \frac{TCL \times \dot{V}_a}{M} \quad (3.51)$$

the relationship between the air quality guideline and the  $TDI_a$  is expressed by

$$TDI_a = \frac{AQG \times \dot{V}_a}{M} \quad (3.52)$$

where  $\dot{V}_a$  represents the human respiratory volume and  $M$  the average human body weight. If neither the TCL nor the air quality guideline are available the inverse of the TDI or the ADI can be used. That is:

$$TDI_a = TDI \quad (3.53)$$

and



$$TDI_a = ADI \tag{3.54}$$

The provisional effect factor for air,  $E_a$ , is then (in order of preference)

$$E_a = \frac{1}{TDI_a} = \begin{cases} \frac{M}{TCL \times \dot{V}_a} \\ \frac{M}{AQG \times \dot{V}_a} \\ \frac{1}{TDI} \\ \frac{1}{ADI} \end{cases} \tag{3.55}$$

where:

- $E_a$  = provisional effect factor for air (day·kg body weight·kg<sup>-1</sup> substance);
- $TCL$  = acceptable concentration in air (kg substance·m<sup>-3</sup> air);
- $AQG$  = air quality guideline (kg substance·m<sup>-3</sup> air);
- $TDI$  = tolerable daily intake (kg substance·day<sup>-1</sup>·kg<sup>-1</sup> body weight);
- $ADI$  = acceptable daily intake (kg substance·day<sup>-1</sup>·kg<sup>-1</sup> body weight);
- $\dot{V}_a$  = human respiratory volume (= 20 m<sup>3</sup> air·day<sup>-1</sup>·person<sup>-1</sup>);
- $M$  = human body weight (= 70 kg body weight·person<sup>-1</sup>).

This can be interpreted as the quantity of body weight which is just exposed to the TCL at a daily intake of 1 kg of the substance via the respiratory tract.

*Development: provisional classification factor for air*

The provisional classification factor is the product of the provisional exposure factor and the provisional effect factor. Hence the provisional human toxicological classification factor for air (HCA) is

$$HCA = X_a \times E_a = \frac{\dot{V}_a \times W \times M}{V_a \times (ACA \text{ or } AQG) \times \dot{V}_a} \text{ or } \frac{\dot{V}_a \times W}{V_a \times (TDI \text{ or } ADI)} \tag{3.56}$$

where:

- $HCA$  = provisional classification factor for air (kg body weight·kg<sup>-1</sup> substance);
- $\dot{V}_a$  = respiratory volume (= 20 m<sup>3</sup> air·day<sup>-1</sup>·person<sup>-1</sup>);
- $W$  = world population (= 5·10<sup>9</sup> persons);
- $M$  = human body weight (= 70 kg body weight·person<sup>-1</sup>);
- $V_a$  = air volume in the model world (= 3·10<sup>18</sup> m<sup>3</sup>);
- $TCL$  = acceptable concentration in air (kg substance·m<sup>-3</sup> air);
- $AQG$  = air quality guideline (kg substance·m<sup>-3</sup> air);
- $TDI$  = tolerable daily intake (kg substance·day<sup>-1</sup>·kg<sup>-1</sup> body weight);
- $ADI$  = acceptable daily intake (kg substance·day<sup>-1</sup>·kg<sup>-1</sup> body weight);

*Development: provisional exposure factor for water*

A similar method to that used for the calculation of the provisional exposure factor for air is used to calculate that an emission into water  $m_w$  (in kg substance) will result in a concentration in water  $C_{in w}$  (in kg substance·litre<sup>-1</sup>) of

$$C_{in w} = \frac{m_w}{V_w} \tag{3.57}$$

where  $V_w$  (in litres) is the quantity of water in the model world. The daily dose taken in with the water  $\dot{m}_{in w}$  (in kg substance·day<sup>-1</sup>·person<sup>-1</sup>) depends on the concentration in water via the volume of the daily water consumption  $\dot{V}_w$  (in l water·day<sup>-1</sup>·person<sup>-1</sup>):

$$\dot{m}_{\text{via } w} = C_{\text{in } w} \times \dot{V}_w = \frac{m_w \times \dot{V}_w}{V_w} \quad (3.58)$$

The total daily human exposure  $T_{\text{via } w}$  (in kg substance·day<sup>-1</sup>) due to a given emission into water is

$$T_{\text{via } w} = \dot{m}_{\text{via } w} \times W = \frac{m_w \times \dot{V}_w \times W}{V_w} \quad (3.59)$$

By dividing  $T_{\text{via } w}$  by the emission  $m_w$  we obtain the *provisional exposure factor for water*  $X_w$  (in day<sup>-1</sup>). This amounts to

$$X_w = \frac{T_{\text{via } w}}{m_w} = \frac{\dot{V}_w \times W}{V_w} \quad (3.60)$$

where:

$X_w$  = provisional exposure factor for water (day<sup>-1</sup>);

$\dot{V}_w$  = human water consumption (= 2 l water·day<sup>-1</sup>·person<sup>-1</sup>);

$W$  = world population (= 5·10<sup>9</sup> persons);

$V_w$  = volume of water in the model world (= 3.5·10<sup>18</sup> l).

Given current assumptions the provisional exposure factor for water is 2.9·10<sup>-9</sup> day<sup>-1</sup>. This can be interpreted as the fraction of an emission into water which the entire world population would be exposed to per day.

*Development: provisional effect factor for water*

The acceptable daily oral intake of a substance, quantified as an ADI or TDI, is inversely proportional to the extent of the toxic effect after oral intake of that substance. Hence the inverse of this value will be proportional to the toxic effect and can be used as the provisional effect factor for intake through drinking water. Hence the provisional effect factor for water  $E_w$  is

$$E_w = \frac{1}{TDI \text{ or } ADI} \quad (3.61)$$

where:

$E_w$  = provisional effect factor for water (day·kg body weight·kg<sup>-1</sup> substance);

$TDI$  = tolerable daily intake (kg substance·day<sup>-1</sup>·kg<sup>-1</sup> body weight).

$ADI$  = acceptable daily intake (kg substance·day<sup>-1</sup>·kg<sup>-1</sup> body weight);

*Development: provisional classification factor for water*

The same method applied to the provisional classification factor for air can be used to calculate the provisional human toxicological classification factor for water (HCW):

$$HCW = X_w \times E_w = \frac{\dot{V}_w \times W}{V_w \times (TDI \text{ or } ADI)} \quad (3.62)$$

where:

$HCW$  = provisional classification factor for water (kg body weight·kg<sup>-1</sup> substance);

$\dot{V}_w$  = human water consumption (= 2 l water·day<sup>-1</sup>·person<sup>-1</sup>);

$W$  = world population (= 5·10<sup>9</sup> persons);

$V_w$  = volume of water in the model world (= 3.5·10<sup>18</sup> l);

$TDI$  = tolerable daily intake (kg substance·day<sup>-1</sup>·kg<sup>-1</sup> body weight).

$ADI$  = acceptable daily intake (kg substance·day<sup>-1</sup>·kg<sup>-1</sup> body weight);

*Development: provisional exposure factor for soil*

A similar method to that used for the provisional exposure factors for air and water can be used to calculate that an emission into the soil  $m_s$  (in kg substance) will result in a concentration in the soil  $C_{\text{in } s}$  (in kg substance·kg soil<sup>-1</sup>) of

$$C_{in,s} = \frac{m_s}{V_s} \quad (3.63)$$

where  $V_s$  represents the quantity of soil (in kg) in the model world\*.

Substances in the soil are generally not taken up by direct consumption of soil but indirectly. The relevant intake routes and the magnitude of the resulting intakes have been modelled by the RIVM (Van den Berg, 1991). As a result the *provisional human toxicological C values* (in kg substance·kg<sup>-1</sup> soil) for a number of substances were developed on the basis of TDI values and data about the relationship between the concentration of each substance in the soil and the daily intake of each of those substances. A human toxicological C value is a measure of the concentration of a substance in the soil which, if exceeded, poses a "serious threat to public health"†.

In essence the C value consists of two components: a component  $p$  to model the extent of exposure per substance and a component which is a measure of the toxicological effect of that substance: the TDI corrected by a factor  $N$ . The C value is proportional to the corrected TDI and inversely proportional to  $p$ :

$$C \text{ value} = \frac{TDI \times N}{p} \quad (3.64)$$

In the publication by Van den Berg the extent of exposure  $p$  (in kg soil·kg<sup>-1</sup> body weight·day<sup>-1</sup>) is not distinguished as such, but using the above formula it can be derived from the C value. In this way the C value is corrected to reverse the uncertainty reduction  $N$  which has now been superseded,  $N$  is eliminated from the C value by division.

In the same way that the daily respiratory volume  $\dot{V}_a$  and the daily water consumption  $\dot{V}_w$  are a measure of the exposure through air and water,  $p$  is a measure of exposure through the soil. The difference is that  $\dot{V}_a$  and  $\dot{V}_w$  relate to human consumption by a person, while  $p$  is expressed per kg body weight. To provide a  $\dot{V}_s$  for soil comparable to  $\dot{V}_a$  and  $\dot{V}_w$ ,  $p$  has to be multiplied by the body weight  $M$  of one person:

$$\dot{V}_s = p \times M = \frac{TDI \times N \times M}{C \text{ value}} \quad (3.65)$$

$\dot{V}_s$  is expressed in kg soil·day<sup>-1</sup>·person<sup>-1</sup>. As exposure to substances emitted to the soil is generally indirect  $\dot{V}_s$  is more difficult to interpret than  $\dot{V}_a$  and  $\dot{V}_w$ .  $\dot{V}_s$  could be interpreted as the fictitious quantity of soil consumed. In other words, the quantity of soil which man is exposed to, directly or indirectly, every day as a result of a given emission. Unlike  $\dot{V}_a$  and  $\dot{V}_w$ ,  $\dot{V}_s$  is not the same for all substances. This is because this factor includes not only the direct exposure but indirect exposure as well. The extent of indirect exposure will differ for each substance, as it depends on the substance properties. For example, a person will only be exposed to the lower soil strata in his garden if it contains substances which can migrate through plastic water pipes and enter the drinking water.

The daily dose via the soil  $\dot{m}_{via,s}$  (in kg substance·day<sup>-1</sup>·person<sup>-1</sup>) of an emitted substance can be calculated using the concentration of that substance in the soil  $C_{in,s}$ , and the daily quantity of "consumed" soil  $\dot{V}_s$  for that substance:

$$\dot{m}_{via,s} = C_{in,s} \times \dot{V}_s = \frac{m_s \times TDI \times N \times M}{V_s \times C \text{ value}} \quad (3.66)$$

The total daily human exposure  $H_{via,s}$  (in kg substance·day<sup>-1</sup>) due to a given emission is

\* To keep the notation consistent the quantity of soil in the model world is referred to as  $V_s$ , however it is not a volume, but a mass.

† Although the assumptions used for the calculation of the C value do not correspond fully with the ideal situation for an LCA (e.g. that only 10% of crops eaten by man are grown in contaminated soil) the C value will be used here as an initial approximation of the human toxicity of emissions into the soil. A more specific approach to human exposure due to emissions into the soil will be investigated at a later stage.



$$T_{\text{via } s} = m_{\text{via } s} \times W = \frac{m_s \times TDI \times N \times M \times W}{V_s \times C \text{ value}} \quad (3.67)$$

The provisional daily exposure factor for soil,  $X_s$  (in  $\text{day}^{-1}$ ), is obtained by dividing  $T_{\text{via } s}$  by the emission  $m_s$ . This amounts to

$$X_s = \frac{TDI \times N \times M \times W}{V_s \times C \text{ value}} \quad (3.68)$$

where:

$X_s$  = provisional exposure factor for soil ( $\text{day}^{-1}$ );

$TDI$  = tolerable daily intake ( $\text{kg substance} \cdot \text{day}^{-1} \cdot \text{kg}^{-1}$  body weight);

$N$  = the uncertainty reduction factor for TDI;

$M$  = human body weight (= 70 kg body weight  $\cdot$  person $^{-1}$ );

$W$  = world population (=  $5 \cdot 10^9$  persons);

$V_s$  = soil mass in the model world (=  $2.7 \cdot 10^{16}$  kg dry matter);

$C \text{ value}$  = human toxicological intervention value ( $\text{kg substance} \cdot \text{kg}^{-1}$  soil).

#### *Development: provisional effect factor for soil*

In the same way that the effect factor for oral intake through drinking water or other water is calculated, we can calculate the provisional effect factor for direct or indirect intake through the soil,  $E_s$ , as the inverse of the TDI:

$$E_s = \frac{1}{TDI} \quad (3.69)$$

where:

$E_s$  = provisional effect factor for soil ( $\text{day} \cdot \text{kg body weight} \cdot \text{kg}^{-1}$  substance);

$TDI$  = tolerable daily intake ( $\text{kg substance} \cdot \text{day}^{-1} \cdot \text{kg}^{-1}$  body weight).

The ADI cannot be used for substances for which there is no TDI as human toxicological C values are not defined for substances for which there is no TDI. Hence it is not possible to calculate a provisional exposure factor for these substances.

#### *Development: provisional classification factor for soil*

The provisional human toxicological classification factor for soil (HCS) is calculated with

$$HCS = X_s \times E_s = \frac{M \times W \times N}{V_s \times C \text{ value}} \quad (3.70)$$

where:

$HCS$  = provisional classification factor for soil ( $\text{kg body weight} \cdot \text{kg}^{-1}$  substance);

$M$  = human body weight (= 70 kg body weight);

$W$  = world population (=  $5 \cdot 10^9$  persons);

$N$  = uncertainty factor of the TDI;

$V_s$  = soil mass in the model world (=  $2.7 \cdot 10^{16}$  kg dry matter);

$C \text{ value}$  = human toxicological intervention value ( $\text{kg substance} \cdot \text{kg}^{-1}$  soil).

#### *Calculation of the effect score*

The provisional human toxicological classification factors for the environmental media air, water and soil are listed in a table in an appendix to the guide. The sources for the toxicity data were: Vermeire *et al.* (1991), FAO/WHO (1990), Staarink & Hakkenbrak (1985 and 1987), WHO (1987), Kleijn & Van der Voet (1991), Van den Berg (1991) and Van den Berg & Roels (1991).

When an LCA is carried out in practice the effect score of each substance is calculated by multiplying the functional unit's emissions into the various environmental media by the relevant provisional or final classification factors. All the effect scores for emissions into the air, water and soil can be added together. This will result in a total effect score for *human toxicity*:

$$\text{human toxicity} = \sum_i ((HCA_i \times m_{a,i}) + (HCW_i \times m_{w,i}) + (HCS_i \times m_{s,i})) \quad (3.71)$$

where:

*human toxicity* = quantity of contaminated body weight (kg body weight);

$m_{a,i}$  = emission into the air (kg substance *i*);

$m_{w,i}$  = emission into water (kg substance *i*);

$m_{s,i}$  = emission into the soil (kg substance *i*);

$HCA_i$  = provisional human toxicological classification factor for air (kg body weight·kg<sup>-1</sup> substance *i*);

$HCW_i$  = provisional human toxicological classification factor for water (kg body weight·kg<sup>-1</sup> substance *i*);

$HCS_i$  = provisional human toxicological classification factor for soil (kg body weight·kg<sup>-1</sup> substance *i*);

This can be interpreted as the total human body weight contaminated up to the maximum acceptable limit as a result of the functional unit. It should be stressed that this effect score should be considered as no more than an indication given the use of provisional classification factors. A more accurate approach will be feasible once the intended model for the classification of toxic substances has been implemented (Guinée & Heijungs, 1992).

### 3.3.4 Ecotoxicity

During the classification of ecotoxic emissions a distinction is made between toxicity to aquatic ecosystems and toxicity to terrestrial ecosystems. No distinction is made between fresh water and salt water ecosystems. For the time being emissions to groundwater and sediment will be included under aquatic and terrestrial ecotoxicity as emissions to these media are almost always indirect and cannot be seen in isolation from environmental processes (which are not considered here). Similarly there is very little data available about the ecotoxic effect of substances taken up through the atmosphere. For this reason toxic substance emissions to the atmosphere will only be assessed for their human toxicological effects (see §3.3.3).

#### *Principles: the provisional exposure factor*

Ecotoxicological exposure factors can be calculated more simply than human toxicological exposure factors. This is because the available toxicity data represent the direct relationship between the concentration of a substance in the environment and its potential ecotoxicological effect. Hence concentration in an environmental medium does not have to be correlated to daily intake through that medium, as is the case in the calculation of the human exposure factor. As a result the ecotoxicological exposure factor depends only on the partition, degradation and immobilization processes in the environment. At the time when this report was published these processes could not be quantified sufficiently. For this reason the *provisional exposure factors*  $X_a$  and  $X_t$  for aquatic and terrestrial ecosystems have been set to 1.

#### *Principles: the provisional effect factor*

A further complication arises when deriving ecotoxicological effect factors. This is that, unlike for human toxicity, the toxic effects for many species have to be combined in a single parameter. An estimate of the effects on an entire ecosystem have to be estimated on the basis of the available toxicity data for a relatively limited number of species. In principle this situation is similar to the derivation of human toxicity data from animal experiments. However, this time the experimental data have to be extrapolated to an ecosystem rather than to mankind. Thus, the extrapolation methods in both cases are similar.

The RIVM has developed a method to assess the ecotoxic effects of substances taken up through water and soil using *maximum tolerable concentrations* (MTCs) (Slooff, 1992). This method is based on two other methods, known as the refined and the preliminary methods. Both are based on the protection of at least 95% of the individuals in the water or soil ecosystem, based on standards for individual substances.

The refined method is based on a proposal by Van Straalen & Denneman (1989), later modified by Aldenberg & Slob (1991). The advantage of this method is that the critical concentration of a substance at which 95% of the organisms are protected can be approximated reasonably well. However, the disadvantage is that this method cannot be widely applied as data for many substances is not available.

The provisional method is the *EPA method* (EPA, 1984) with some modifications made by the RIVM (Van de Meent *et al.*, 1990). This method can only provide a coarse approximation of the critical concentration at which 95% of individuals in an ecosystem are protected. The coarse approach is more conservative than the refined one as the risk of an error on the wrong side of the limit (less than 95% protected) is greater when the coarse approach is used than when the refined approach is used. The average safety margin is higher to be on the safe side. The choice of safety margin depends on the volume of data available, but it lacks scientific foundation.

Slooff (1992) proposes using either the refined method or the provisional method, depending on the volume of data available for each substance. Naturally the refined method is preferable. The critical concentration determined in this way is the MTC.

As an MTC can be determined in two different ways its value has no direct correlation with the toxicity of a substance. The MTC is relatively low (i.e. conservative) for substances on which little data is available. In other words, the MTCs of different substances cannot be compared directly. As the standards on which the classification is based in an LCA have to be comparable the MTC cannot be used directly for this.

For the time being the EPA method will be used to determine an "MTC" for terrestrial and aquatic ecosystems to make a practical implementation feasible. The "MTCs" derived in this way are identified as  $MTC_{EPA}$ . On this basis separate classification factors can be distinguished for water and soil: the ECA (ecotoxicological classification factor for aquatic ecosystems) and the ECT (ecotoxicological classification factor for terrestrial ecosystems).

#### *Development: provisional classification factors*

Only the provisional classification factor has to be developed as the provisional exposure factor has been set at 1. The ecotoxicological effect factor is derived from toxicity data such as the *no observed effect concentration* (NOEC), the *lethal concentration for 50% of the organisms* ( $LC_{50}$ ) and the *effect concentration for 50% of the organisms* ( $EC_{50}$ ) for individual aquatic or terrestrial species or a QSAR estimate (QSAR = *quantitative structure-activity relationship*) of one of these parameters. The  $MTC_{EPA}$  is derived from this data by multiplying the toxicity data by a safety factor which depends on the volume and quality of the toxicity data available. These safety factors are listed in Tables 3.2 and 3.3 (Slooff, 1992).

TABLE 3.2. Procedure to determine the  $MTC_{EPA}$  for aquatic ecosystems.

data required	extrapolation factor
lowest acute $L(E)C_{50}$ or QSAR estimate of acute toxicity	0.001
lowest acute $L(E)C_{50}$ or QSAR estimate of acute toxicity to at least one representative of three of the four groups <i>algae</i> , <i>crustaceans</i> and <i>fish</i>	0.01
lowest chronic NOEC or QSAR estimate of chronic toxicity to at least one representative of three of the four groups <i>algae</i> , <i>crustaceans</i> and <i>fish</i> <sup>†</sup>	0.1*

\* If fewer than three chronic NOECs were available but acute toxicity data was available, the available data was used to undertake both the last and one of the two procedures at the top of the table. In which event the lowest value obtained was used. If fewer than three chronic NOECs were available and there was no data available on toxicity, the lowest NOEC was multiplied by an extrapolation factor of 0.01 after consultation with Slooff.

† The use of *MicroTox* data is acceptable.

If different but equally authoritative toxicity data based on the same toxicological parameter is



available for a species, the geometric mean of the data is used\*. If the toxicological parameters differ the lowest value is used.

TABLE 3.3. Procedure to determine the  $MTC_{EPA}$  for terrestrial ecosystems.

required data	extrapolation factor
lowest acute $L(E)C_{50}$ or QSAR estimate of acute toxicity	0.001
lowest acute $L(E)C_{50}$ or QSAR estimate for acute toxicity for at least one representative of three of the four groups <i>microbial processes</i> , <i>earthworm</i> , <i>arthropoda</i> and <i>plants</i>	0.01
lowest chronic NOEC or QSAR estimate of chronic toxicity for at least one representative of three of the four groups <i>microbial processes</i> , <i>earthworms</i> , <i>arthropoda</i> and <i>plants</i>	0.1*

\* If fewer than three chronic NOECs were available but acute toxicity data were available the available data was used to undertake the last procedure as well as one of the first two, and the lowest value was used. If fewer than three chronic NOECs were available and there was no data about acute toxicity either the lowest NOEC was multiplied with an extrapolation factor of 0.01, after consultation with Slooff.

The toxicological report in the *Derde Nota Waterhuishouding* (Stortelder *et al.*, 1989) provided toxicity data for aquatic organisms. The "lowest NOECs for unicellular algae, molluscs, crustaceans and fish" are often estimates based on other toxicological data, but are still used as the lowest NOEC for these groups. The appendices to the RIVM report by Van de Meent *et al.* (1990) and Denneman & Van Gestel (1990) provided the toxicity data for terrestrial ecosystems.

The effect factor for aquatic ecosystems,  $E_a$ , is inversely proportional to the maximum acceptable concentration  $MTC_{EPA}$ , and is expressed by

$$E_a = \frac{1}{MTC_{EPA}} \tag{3.72}$$

similarly the effect factor for terrestrial ecosystems can be calculated with

$$E_t = \frac{1}{MTC_{EPA}} \tag{3.73}$$

The ecotoxicological classification factor for aquatic ecosystems (ECA) is

$$ECA = X_a \times E_a = \frac{1}{MTC_{EPA}} \tag{3.74}$$

where:

$ECA$  = provisional ecotoxicological classification factor for aquatic ecosystems ( $m^3$  water· $mg^{-1}$  substance);

$MTC_{EPA}$  = maximum tolerable concentration determined for that medium using the EPA method ( $mg$  substance· $m^{-3}$  water).

The ecotoxicological classification factor for terrestrial ecosystems (ECT) is

$$ECT = X_t \times E_t = \frac{1}{MTC_{EPA}} \tag{3.75}$$

\* The geometric mean  $\Gamma$  is the  $e$ th power of the mean of the natural logarithms of the individual values:

$$\Gamma = \exp \left[ \frac{1}{N} \sum_{i=1}^N \ln a_i \right] = \left[ \prod_{i=1}^N a_i \right]^{1/N}$$

where:

$ECT$  = provisional ecotoxicological classification factor for terrestrial ecosystems (kg soil·mg<sup>-1</sup> substance);

$MTC_{EPA}$  = maximum tolerable concentration, determined for that environmental medium using the EPA method (mg substance·kg<sup>-1</sup> soil).

#### Calculation of the effect scores

The provisional ecotoxicological classification factors for the environmental media water and soil are listed in the tables in the appendix of the guide. During an actual study the effect score of each substance is calculated by multiplying the emissions of the functional unit to the various media by the appropriate provisional or definitive classification factors. The effect score for aquatic ecotoxicity is calculated with

$$\text{aquatic ecotoxicity} = \sum_i ECA_i \times m_{w,i} \quad (3.76)$$

where:

$\text{aquatic ecotoxicity}$  = volume of the polluted aquatic ecosystem (m<sup>3</sup> water);

$m_{w,i}$  = emissions to water (mg substance  $i$ );

$ECA_i$  = provisional ecotoxicological classification factor for aquatic ecosystems (m<sup>3</sup> water·mg<sup>-1</sup> substance  $i$ ).

The effect score for terrestrial ecotoxicity is calculated with

$$\text{terrestrial ecotoxicity} = \sum_i ECT_i \times m_{t,i} \quad (3.77)$$

where:

$\text{terrestrial ecotoxicity}$  = volume of polluted terrestrial ecosystem (kg soil);

$m_{t,i}$  = emissions to the soil (mg substance  $i$ );

$ECT_i$  = provisional ecotoxicological classification factor for terrestrial ecosystems (kg soil·mg<sup>-1</sup> substance  $i$ ).

The resulting values *terrestrial ecotoxicity* and *aquatic ecotoxicity* are expressed in kg soil and m<sup>3</sup> water. They can be interpreted as the quantity of terrestrial or aquatic ecosystem polluted to the  $MTC_{EPA}$ . Thus in this provisional method the critical volumes approach is used to classify ecotoxicity. Again this effect score should be considered as no more than an indication, given the use of *provisional* exposure factors. A more accurate approximation will be possible once the intended model for the classification of toxic substances has been implemented (Guinée & Heijungs, 1992).

#### 3.3.5 Acidification

The *acidification potential* (AP) of a substance  $i$  is calculated on the basis of the number of H<sup>+</sup> ions which can be produced per mole substance. This is given by the stoichiometric coefficient  $\nu$  in the equation



where  $X$  is the acidifying substance. As environmental interventions are specified in kg emission rather than moles this has to be divided by the molecular weight  $M$  of the substance:

$$\eta (\text{mol} \cdot \text{kg}^{-1}) = \frac{\nu}{M (\text{kg} \cdot \text{mol}^{-1})} \quad (3.79)$$

The *acidification potential* of a substance  $i$  ( $AP_i$ ) is obtained by dividing  $\eta_i$  by  $\eta_{ref}$ . SO<sub>2</sub> is used as the reference substance.

$$AP_i = \frac{\eta_i (\text{mol} \cdot \text{kg}^{-1})}{\eta_{\text{SO}_2} (\text{mol} \cdot \text{kg}^{-1})} \quad (3.80)$$

Table 3.4 lists the values calculated for a range of acidifying substances.

TABLE 3.4. Calculation of the acidification potential of a range of acidifying substances.

compound	reaction equation	$\nu$	$M$ (kg·mol <sup>-1</sup> )	$\eta$ (mol·kg <sup>-1</sup> )	AP
SO <sub>2</sub>	SO <sub>2</sub> + H <sub>2</sub> O + O <sub>3</sub> → 2H <sup>+</sup> + SO <sub>4</sub> <sup>2-</sup> + O <sub>2</sub>	2	64	1/32	1.00
NO	NO + O <sub>3</sub> + ½H <sub>2</sub> O → H <sup>+</sup> + NO <sub>3</sub> <sup>-</sup> + ¾O <sub>2</sub>	1	30	1/30	1.07
NO <sub>2</sub>	NO <sub>2</sub> + ½H <sub>2</sub> O + ¼O <sub>2</sub> → H <sup>+</sup> + NO <sub>3</sub> <sup>-</sup>	1	46	1/46	0.70
NO <sub>x</sub> *	-	1	46	1/46	0.70
NH <sub>3</sub>	NH <sub>3</sub> + 2O <sub>2</sub> → H <sup>+</sup> + NO <sub>3</sub> <sup>-</sup> + H <sub>2</sub> O	1	17	1/17	1.88
HCl	HCl → H <sup>+</sup> + Cl <sup>-</sup>	1	36.5	2/73	0.88
HF	HF → H <sup>+</sup> + F <sup>-</sup>	1	20	1/20	1.60

\* For NO<sub>x</sub> an average value of 2 was used for x.

3.3.6 Nutrification

The calculation of the *nutrification potential* (NP) of a substance *i* is based on the average composition of algae: C<sub>106</sub>H<sub>263</sub>O<sub>110</sub>N<sub>16</sub>P (Eijsackers *et al.*, 1985), which is assumed to be representative of the average composition of biomass. As discussed in the introduction to the concept NP (§3.2.9), only emissions of substances containing nitrogen or phosphorus are considered as nutrififying. For each nutrient X it is determined to what extent it contributes to the formation of biomass, assuming that the supply of other nutrififying substances is unlimited:



When dead biomass is emitted the O<sub>2</sub> required for its degradation can be measured as the chemical oxygen demand (COD). It is assumed that one mole of biomass requires 138 moles O<sub>2</sub>:



If the biological oxygen demand (BOD) is specified it can generally be converted to a COD. The conversion factor will depend on the situation.

In the same way as for acidification, the stoichiometric coefficient  $\nu$  has to be divided by the molecular weight *M* of the substance concerned\*:

$$\eta(\text{mol}\cdot\text{kg}^{-1}) = \frac{\nu}{M(\text{kg}\cdot\text{mol}^{-1})} \tag{3.83}$$

The *nutrification potential* of substance *i* (NP<sub>*i*</sub>) is obtained by dividing  $\eta_i$  by  $\eta_{ref}$ . PO<sub>4</sub><sup>3-</sup> is used as the reference substance.

$$NP_i = \frac{\eta_i(\text{mol}\cdot\text{kg}^{-1})}{\eta_{PO_4^{3-}}(\text{mol}\cdot\text{kg}^{-1})} \tag{3.84}$$

Table 3.5 lists the values calculated for a number of nutrififying substances.

\* Hence nitrogen and phosphorus have to be added in mass ratios of 31 ÷ 224 = 1 ÷ 7.23. This differs from the ratio of 1 ÷ 10 used in the Milieuprogramma 1992-1995 (VROM, 1991a) which was used as a policy indicator on the basis of the emission ratio.



TABLE 3.5. Calculation of the nitrification potential of various nitrifying substances.

compound	$\nu$	$M$ (kg·mol <sup>-1</sup> )	$\eta$ (mol·kg <sup>-1</sup> )	$NP$
N	1/16	14	1/224	0.42
NO	1/16	30	1/480	0.20
NO <sub>2</sub>	1/16	46	1/736	0.13
NO <sub>x</sub> *	1/16	46	1/736	0.13
NO <sub>3</sub> <sup>-</sup>	1/16	62	1/992	0.10
NH <sub>4</sub> <sup>+</sup>	1/16	18	1/288	0.33
P	1	31	1/31	3.06
PO <sub>4</sub> <sup>3-</sup>	1	95	1/95	1.00
COD (as O <sub>2</sub> )	1/138	32	1/4416	0.022

\* For NO<sub>x</sub> an average value of 2 was used for  $x$ .

### 3.4 Discussion

*Our perceptual knowledge of nature consists of the breaking up of a whole which is the subject matter of perceptual experience – or however else we prefer to describe the ultimate experienced fact. This whole is discriminated as being a complex of related entities, each having determinate qualities and relations and being a subject concerning which our perceptions, either directly or indirectly, afford definite information.*

ALFRED NORTH WHITEHEAD, *An enquiry concerning the principles of natural knowledge.*

A first step has been made there to develop classification factors for a wide range of environmental problems. It should be stressed that a number of these classification factors require further development and improvement. The proposals in this document for the classification should not be considered final, they require further development in at least two respects:

- validation, improvement and development of the factors;
- development of the factors for generic classifications at a lower level of scale.

The classification factors proposed above will require validation, improvement, further refinement and regular updating\*. It is extremely important that scientific panels discussion are set up for various classification factors, to coordinate and approve this process. Specialist discussion panels have already been instituted for a number of classification factors such as the GWP, the ODP and the POCP. These panels coordinate and approve the results of international research. For the ODP there is the *Scientific Assessment Panel* under the auspices of the *World Meteorological Organization* (WMO), for the GWP *Working Group I* of the *International Panel on Climate Change* (IPCC) under the auspices of the WMO and the *United Nations Environment Programme* (UNEP), and for the POCP the *Working Group on Volatile Organic Compounds* under the auspices of the *United Nations Economic Commission for Europe* (UNECE). At present toxicity is not yet covered. It would be a good idea to create similar discussion panels for this and other problems such as acidification and nitrification. For example, a future panel on toxicity could discuss the HTP and TETP/AETP concepts suggested by Guinée &

\* The inclusion of Mackay's multi-media dispersal models as used for the HTP, TETP and AETP (Guinée & Heijungs, 1992) might improve some of the classification factors related to substances. For example, at present only atmospheric emissions are assessed for their greenhouse effect. If the same or equivalent substances (e.g. carbon dioxide as carbonate) are emitted to water and evaporate they are not assessed for their greenhouse effect. The addition of a multi-media dispersal model to the current model, similar to that proposed by Mackay could remedy this possible shortcoming in current greenhouse-effect models.

Heijungs (1992)\*. This would result in the development of more uniform and international scientifically approved classification methods.

The other aspect which requires further study concerns the opportunities and obstacles to giving the classification a more spatial dimension. It was claimed earlier that a location specific classification is not feasible in the context of an LCA regardless of the level of scale. However, for certain types of problems (nutrification, acidification, desiccation, etc.) a classification which is more detailed than a generic one at global level may be desirable or even necessary. Dividing the world into ten areas, for example, and determining the generic classification factors for those areas could be a first step towards a spatial differentiation of the classification.

A list of all relevant environmental interventions on which data has to be collected during the classification can be compiled on the basis of the proposed classification factor. This list can be used as a basis for the further development of the format for storing process data in the inventory analysis. This has not been included in this report; the classification factors given in the appendix to the guide provide a basis for this systematic list. The list and structure of the format will have to be updated regularly, as will the actual classification factors. The development of classification factors for a number of environmental interventions has been described. Once these classification factors have been developed the relevant data will have to be collected during the inventory analysis, e.g. data on radiation emissions. Clearly, this will require considerable extra work and a balance will have to be struck between aiming for a comprehensive LCA which includes all the environmental problems, and the costs, work and time this would require.

Information about the life cycle of a product is used to make recommendations for the redesign of a different process operation. Clearly priorities will have to be established: what is more important? Dealing with acidification or with the greenhouse effect? Or, in terms of the inventory analysis, should we reduce NO<sub>x</sub> emissions or cadmium emissions? These types of evaluation (concerning several products or a single product) will have to be considered during the actual evaluation.

The classification will have produced a normalized environmental profile consisting of ten to twenty effect scores. Effect scores can only be compared when product alternatives are being compared. In a few cases it will then be possible to draw a conclusion without further weighting. This is only possible when all the effect scores of a product alternative are better than those of the other product. This form of unweighted comparison was included as the last element in step 3.3 of the classification, as it does not require the essential element of the evaluation, i.e. the rating of preferences.

However, in many cases one product alternative will do better on some effect scores but worse on others. In this event the effect scores will have to be rated in order to make an assessment. At least two methods can be used for this:

- \* qualitative multi-criteria analysis;
- \* quantitative multi-criteria analysis.

These are general terms covering a number of implementations (Jansen, 1991; Ministerie van Financiën, 1986). Both methods include methodological as well as procedural aspects. The two sections below focus on the methodological aspects. The procedural aspects are largely concerned with issues such as "who will undertake the evaluation?" and "what information is provided to those concerned?"

In the qualitative method a person or forum rate the better and poorer effect scores for each case. This rating may well be implicit. This method is not uniform: when a set of two environmental profiles is assessed by two people or forums their assessments may be different.

The methodological aspects of a qualitative MCA are limited<sup>†</sup>. One of the methodological issues is the form in which environmental profiles should be submitted for assessment. This concerns to the explicit or implicit manner by which the rating is done. The assessment might be based on the

<sup>†</sup> In addition to the term multi-criteria analysis (MCA) terms such as multi-criteria method and multi-criteria evaluation are also used.

\* We suggested creating a toxicity panel of this kind at a symposium on LCA held in Potsdam. The response was encouraging and Professor Bro-Rasmussen of the *Technical University of Denmark* is currently working on the introduction of such a panel.

## CHAPTER 4

# EVALUATION

The actual assessment of the environmental profile of the product or products under investigation takes place during the evaluation. The nature of the assessment is determined by the type of application. Usually this will be a comparative assessment. Other examples include providing information about a product range with the same function, product regulation by government and comparing a product with one or more redesigned versions of that product.

The assessment is only an independent entity when the aim is to find options to improve a product. During the improvement analysis information about the life cycle of a product is used to make recommendations for the redesign or a different process operation. Clearly priorities will have to be established: what is more important? Dealing with acidification or with the greenhouse effect? Or, in terms of the inventory analysis, should we reduce NO<sub>x</sub> emissions or cadmium emissions? These types of evaluation (concerning several products or a single product) will have to be considered during the actual evaluation.

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\* In addition to the term multi-criteria analysis (MCA) terms such as multi-criteria method and multi-criteria evaluation are also used.

† For this reason some claim that it should not be considered as a method, but only as a procedure.



contribution made to current environmental problems. In this event the normalized environmental profile created in step 3.4 could be used. Another option would be to add other information to the environmental profiles, e.g. by comparing them with levels of sustainability. This will be discussed in greater detail during the development of the methodological aspects of the qualitative MCA (§4.1). To deal with the procedural aspects requirements could be defined with regard to the membership and the procedures of the expert forum. These requirements could be included in an LCA code of conduct. These aspects will not be discussed in this report.

A quantitative MCA is based on weighting factors used for the explicit weighting of the effect scores. Such an MCA consists of two parts. The weighting factors are defined in the first part. This may be done separately for each assessment. Naturally a standard could be developed, not based on any given study but rather on more abstract information. The weighting standards defined in such a standard can then be applied in each study to rate the effect scores. In this way the structure of the evaluation will resemble that of the classification: an expert committee would publish a list of evaluation factors which could be used in a simple formula during a product study.

At present there is no such list of weighting factors, or even a national or international expert committee. Therefore the quantitative MCA has not been included in the guide. This chapter includes some of the options for its implementation.

Alternatives to the above methods are also possible. For example in the *hierarchical exclusion method* the user gives absolute priority to one environmental effect. In this case a product which does better in that aspect will always be rated higher than the other product. If the environmental profiles for the aspect concerned are identical then a second criterion will be considered. When this evaluation method is used it is more efficient to limit the classification to the one or two most important environmental effects, rather than covering the complete environmental profile. Many consider this approach as unsophisticated as it does not make good use of a comprehensive LCA with detailed quantitative information. The method overlooks more subtle considerations, for example many slightly harmful emissions may be worse than a single very harmful emission. Its use will often require a different classification method as exclusion criteria will not have been defined for acidification, etc., but criteria such as "contains a black-listed substance" or "contains tropical hardwood" will have been included. In the discussion of the evaluation this method will not be considered for use in LCAs. It is only mentioned here as it can be used for rapid evaluations for informative purposes, such as described in Elkington & Hailes (1989). The second and third alternative methods are multi-criteria analyses (MCA).

The assessment of an environmental profile depends on its reliability. Uncertainty about the effect scores can make an assessment useless, irrespective of whether it is a formalized or an *ad hoc* appraisal. Thus the evaluation also includes an appraisal of the reliability of the environmental profile.

Therefore the evaluation is divided in two steps in the guide:

- evaluation of the environmental profile (step 4.1);
- evaluation of the reliability and validity (step 4.2).

In this background document the technical requirements will be developed for the quantitative method in step 4.1. The procedural aspects are more suitable for negotiations within the framework of a code of conduct. However, the technical requirements will provide some recommendations for the questions to be addressed in such a procedure.

## 4.1 Quantitative multi-criteria analysis

This section deals with the technical aspects of a quantitative multi-criteria analysis (MCA). Both the creation of the environmental index\* and the definition of the weighting factors will be discussed. The

\* The term *environmental index* is used here without considering other nomenclature. As far as we are aware there are no other names for this quantity. The method proposed by Ahbe *et al.* (1990) leads to an assessment in which the overall effect of a product on the environment is expressed in Öko-Punkte. This defines Öko-Punkte as a unit, albeit an unofficial one (see e.g. Klein, 1988). The quantity expressed in this way does not have a name. Alternative names include *environmental*

definition of the weighting factors will be discussed only in terms of the technical requirements.

#### 4.1.1 Definition of the environmental index

In a quantitative MCA  $m$  effect scores  $\gamma_i$  ( $i = 1, \dots, m$ ) are multiplied by  $m$  weighting factors  $\mu_i$  ( $i = 1, \dots, m$ ) and the results are totalled to create the environmental index  $M$ :

$$M = \sum_{i=1}^m \mu_i \gamma_i \quad (4.1)$$

The only formal requirement for the weighting factors  $\mu_i$  is that their dimension  $[\mu_i]$  should be such that

$$[\mu_i] = \frac{[M]}{[\gamma_i]} \quad (4.2)$$

A number of environmental indexes could be created by choice or necessity. The theory behind multiple environmental indexes can easily be derived from this\*. Discussion of the quantitative MCA will focus on single environmental indexes.

#### 4.1.2 Definition of the weighting factors

In practice the formal method based on weighting with weighting factors  $\mu_i$  of different dimensions is difficult to apply. This is because the definition is based on statements such as "1 mol H<sup>+</sup> is twice as bad as 1 gramme CO<sub>2</sub> equivalent", rather than "acidification is twice as bad as the greenhouse effect". Statements such as these cannot be made without an appreciation of the significance of different units.

To determine the weighting factors properly their definition should be based on an abstract approach rather than on an actual example. A sample product could be used for appraisal purposes, but if it is used to derive the weighting factors it is difficult to ignore prejudices about the product.

Four methods which can be used in the definition of weighting factors will be discussed here (see also Sas & Sauer, 1992):

- weighting using monetary methods;
- weighting by societal preferences;
- application of levels of sustainability;
- modelling of the eventual effects.

Some of these methods are partly based on the natural sciences, all four contain a social scientific element. As a consequence part of the evaluation may have to be carried out during the classification.

##### *Weighting using monetary methods*

In a monetary method the effect scores are associated with a price. A monetary measure of the harmfulness of a product is obtained by multiplying each effect score by the price per effect score unit and totalling these results. Two forms of monetary valuation can be distinguished.

The first method is based on the effects-oriented policy: the damage is expressed in monetary terms (Hueting *et al.*, 1992). The disadvantage of this method is that the effects have to be determined at a later stage in the causal chain than that studied in the classification. Hoevenagel and Opschoor (1990) have indicated when the monetary damage can be determined: when the complete effect occurs in the short term. This is not relevant for an LCA.

score or environmental indicator.

\* The environmental indexes  $M_j$  can be defined as

$$M_j = \sum_{i=1}^m \mu_{ji} \gamma_i$$

where  $\mu_{ji}$  represents the weighting factor for environmental index  $M_j$  and effect score  $\gamma_i$ . For example, indexes for depletion, pollution and damage could be defined. In this event the weighting factor for the  $i^{\text{th}}$  effect score would be zero for two of the three indexes: each effect score contributes to just one environmental index. However, this is not a universal requirement. In theory environmental indexes could have different units.



The second method is related to the source-oriented policy: the costs required to prevent the effects are expressed in monetary terms. The DESC developed by TEBODIN (Krozer, 1990) is an example of this method. It is used to calculate the cheapest method of achieving a defined emission reduction. These costs will differ for each substance and in each situation.

One of the disadvantages of the monetary approach is that it depends on the level of technological development. Certain relatively harmful effects can only be avoided at high cost. A further disadvantage is that the cost required to avoid an emission bears no relation to the seriousness of that emission\*. Furthermore it is almost impossible to quantify depletion and damage.

#### *Weighting by societal preferences*

In this method the opinion of an individual or group of individuals is surveyed†. The weighting factors can be set by scientists, politicians, a broadly representative forum, plebiscite or survey (*stated preference*). Alternatively the weighting factors could be reconstructed on the basis of recent policy decisions (*revealed preference*). When the problems can only be placed in order, an extreme weights method can be used to change the weights within the priorities set and determine the effects on the outcome.

In a recent study a panel of representatives from science, industry and the environmental movement were asked to determine the relative weights of a number of environmental problems (McKinsey & Company, 1991). The grounds on which these ratings were based would be an interesting subject of social scientific study. The influence of the questions on the result is also uncertain. For example, let us consider the question "Is the greenhouse effect more important than acidification, and if so by how much?". We will assume that this question elicits a positive response, the quantification given is 10 times and the dispersion is relatively small. As the greenhouse effect and acidification are not quantified in the question it may be concluded that these experts do not refer to absolute numbers. Clearly they do not mean that 1 g CO<sub>2</sub> is ten times worse than 1 mol H<sup>+</sup> or that a 1 m rise in the sea level is ten times as bad as 1 ha dead forest. It is likely that those surveyed based their statement on their understanding of scientific, possibly popular scientific, reports on the consequences of the effects *which are occurring now or are predicted*, including all the uncertainties surrounding this‡. For this reason the statement "the greenhouse effect is ten times as important as acidification" has to be considered in the context of the total magnitude of the greenhouse effect and acidification in the world over a defined period§. To be able to use the weighting factors correctly the same units as those used in the classification would have to be used here. This can be done as follows: if the dimensionless "significance factor" for the *l*<sup>th</sup> effect score is represented by  $\psi_l$  and the annual magnitude of the problem by  $\dot{\gamma}_{w,l}$ , the corresponding weighting factor  $\mu_l$  can be given by

$$\mu_l = \frac{\psi_l}{\dot{\gamma}_{w,l}} \quad (4.3)$$

As a result the environmental index is expressed in time<sup>¶</sup>.

The following considerations may be relevant when assessing the significance of a problem:

- the current magnitude of the problem;
- the current annual increase in the problem;
- the predicted magnitude of the problem;
- the difficulty of controlling the problem;

\* The environmental annual report of BSO/ORIGEN (1991) states: "This approach is based on the assumption that society takes a rational view of environmental issues, i.e. that environmental protection measures are taken until the marginal costs of those measures equal the marginal benefits."

† This is the only method which could be developed on the basis of an actual product example. However we advise against this for the reasons given earlier.

‡ This would mean that the current scale would also have to be used in historical product assessments.

§ If that information is unavailable this could be limited to the extent of the problem in the Netherlands. This would have to be done consistently for all effect scores.

¶ Time in this expression should not be interpreted as an actual period.



- the significance of the problem in the most extreme situation;
- etc.

Hence, even when this approach is used the questions have to be drafted extremely carefully.

#### *Application of levels of sustainability*

When levels of sustainability (or notions such as environmental capacities) are used a weighting factor is defined for each effect score based on an accepted level of the effect. Different levels of sustainability can be distinguished. Levels of sustainability which are only based on environmental considerations are best, but they are also the most difficult. Furthermore, there are policy levels of sustainability which are a long-term compromise between what is desirable on environmental grounds, what is technically possible and what is economically feasible. There are also intended policy levels, i.e. short-term compromises based on the same considerations\*. The Swiss eco-points system referred to earlier (Ahbe *et al.*, 1990) is based on levels of sustainability set by politicians.

Environmental sustainability levels are preferred as they are more objective than policy levels. It is uncertain, however, to what extent environmental sustainability levels can be determined. Furthermore there is the question whether exceeding the sustainability level for the greenhouse effect by a factor two is as serious as exceeding the acidification level by the same factor. This will depend on the way in which the sustainability levels are defined. The sustainability level can be defined as the maximum effect during a given period: hence its dimension is the same as that of the magnitude of the effect†. The sustainability level of the  $l^{\text{th}}$  effect is represented by  $\dot{\gamma}_{s,l}$ . As above the weighting of the effect scores is calculated with

$$M = \sum_{l=1}^m \psi_l \frac{\gamma_l}{\dot{\gamma}_{s,l}} \quad (4.4)$$

The factor  $1/\dot{\gamma}_{s,l}$  is known as the *contribution factor* and the factor  $\psi_l$  the *weighting factor* (Sas & Sauer, 1992). The magnitude of the weighting factor (which indicates the significance of exceeding the sustainability level) depends on the magnitude of the sustainability level which partly depends on the definition of "sustainable". In principle the level of sustainability could be set such that the weighting factor  $\psi_l$  is 1 for all effect scores, in which case the level of sustainability  $\dot{\gamma}_{s,l}$  is defined such that exceeding this level by a given amount will be equally significant for all effects. This illustrates the flexible nature of the term sustainable. The level of sustainability for the greenhouse effect could be set at 1 cm or 1 mm rise in the sea level per century. This information should be available when the weighting factor is set: the factor will have to vary by a factor of 10. To stress the subjective aspects it would be better to use the term acceptable level‡ rather than level of sustainability.

#### *Modelling the eventual effects*

The last parameter to be discussed here is environmental damage, expressed in terms other than economic. An attempt could be made to complete the last part of the effect chain (which represents the correlation between intervention, first order effect, second order effect, etc.) by modelling the last part of this chain in order to determine the damage. In this way the environmental effects are plotted against a target variable representing the condition of society (similar to Figure 0.5). Potential target variables include human health and safety, prosperity, welfare and intrinsic natural values (Udo de Haes, 1991). Quite apart from the question whether it would be better to base the rating on the perception of the significance by society, it has to be asked whether all four target variables should

\* For example: the NEPP lists a number of levels. These can be interpreted as follows: environmental sustainability level: 400-700 acid eq/ha/yr; policy sustainability level: 1400 acid eq/ha/yr; intended policy level: 2400 acid eq/ha/yr.

† Again, this will result in an expression with the dimension time. In this event the number can be interpreted as a period: it is the period during which the earth is exposed to all environmental problems caused by the product up to the set limits.

‡ This recurs when defining a level of sustainability for the depletion of resources. As these will eventually run out the actual level of sustainability should be zero and almost every product would have an infinitely high environmental index. In practice an acceptable level, other than zero, can be defined for resources.

be included in the evaluation, just one or if they should be combined. This would provide a direct rating of the environmental effects.

There are some similarities between this method and levels of environmental sustainability. When this approach is used a sustainable level is defined for each effect. The extent to which that level is occupied is aggregated across the effects. Levels of sustainability are defined in terms of a given effect per year. This could be linked to the way in which the effects are defined in the classification of an LCA. For example the level of sustainability for acidification could be set at  $10^9 \text{ kg}\cdot\text{yr}^{-1}$ . The effect score can be linked to this by division, resulting in a number with the dimension time. When modelling the eventual effects a common target variable (e.g. human health) is formulated and all effect scores are related to this meta-effect by means of an evaluation factor, similar to the classification factor. However, defining evaluation factors by predicting the target variables may well be even more complicated than defining scientifically sound levels of sustainability.

It will be clear from the above considerations that the definition of weighting factors will also include a normative element. This provides further justification for the distinction between the natural sciences component, i.e. the classification, and the social-scientific component, i.e. the evaluation.

## 4.2 Sensitivity analysis

The reliability of the environmental profiles or the inventory tables affects the certainty with which a conclusion can be drawn. The term *sensitivity analysis* is commonly used to refer to the determination of the influence of changes on the end results. Four types of sensitivity analysis are distinguished in this study:

- reliability analysis;
- validity analysis;
- dominance analysis
- marginal analysis.

Dominance analysis\* and marginal analysis are discussed in the chapter on improvement analysis (Chapter 5). Reliability analysis and validity analysis will be discussed here. There will be some overlap as the marginal analysis can also be used for reliability analysis.

### 4.2.1 Reliability analysis

Reliability analysis is the analysis of data when the dispersion of the process data is specified or can be estimated. This results in an inventory table, environmental profile or environmental index and a specification of the dispersion. It is difficult to make a comparative judgement if the environmental index of one product lies within the dispersion of another environmental index.

Almost all relevant data in an LCA suffer from some inaccuracies. The highest or expected inaccuracy of some of this data can be determined on the basis of a data series or estimates. This can be specified as a spread (a parameter has a value of 8 with a lower limit of 5 and an upper limit of 10) or by specifying the error (a parameter has a value of  $14 \pm 3$ ).

Error analysis can be used to estimate the error in the result of a calculation due to errors in the data. Heijungs (1992) has developed a method to calculate the estimated error in inventory analysis calculations. The magnitude of the  $k^{\text{th}}$  intervention  $\beta_k$  is found by adding the intervention  $b_{ki}$  to all processes in proportion to the occurrence of  $p_i$  in those processes (see §2.4.1):

$$\beta_k = \sum_{i=1}^q b_{ki} p_i \quad (4.5)$$

For the error  $\Delta\beta_k$  due to the errors  $\Delta b_{ki}$  and  $\Delta p_i$  the following applies up to the first order:

\* Strictly speaking a dominance analysis (§5.1) is not a form of sensitivity analysis as no data is varied and the sensitivity to variation is not assessed.

$$\Delta\beta_k = \sum_{i=1}^q (b_{ki}\Delta p_i + p_i\Delta b_{ki}) \quad (4.6)$$

where  $\Delta b_{ki}$  is a measured or estimated parameter and  $\Delta p_i$  is calculated as follows:

$$\Delta p_i = \sum_{j=1}^r \left[ \left[ -P_i \sum_{i=1}^q \frac{(-1)^{j+i} \det(A_{ji})}{\det(A)} \Delta a_{ji} \right] + \left[ \sum_{i=1}^q \frac{(-1)^{j+i} \det(A_{ji}^t)}{\det(A)} \Delta a_{ji} \right] + \left[ \frac{(-1)^{j+i} \det(A_{ji})}{\det(A)} \Delta \alpha_j \right] \right] \quad (4.7)$$

where  $\det(A_{ji})$  represents the *minor* of the process data matrix  $A$ : the determinant of matrix  $A$  in which the  $j^{\text{th}}$  row and  $i^{\text{th}}$  column have been eliminated. In this way each effect score can be written as  $\beta_k \pm \Delta\beta_k$ .

This systematic approach cannot be used for process data specified with an upper and a lower limit instead of a dispersion. Of course, all process data could be varied from the lower to the upper limit, but it is less cumbersome to define a dispersion which includes both the lower and the upper limit to a reasonable extent.

If the dispersion of the process data is unknown it could be assumed that the dispersion is zero or almost zero, or that the process data is fairly accurate. This is indeed valid for some process data: a bike will be fitted with exactly one bell, a crate accommodates exactly 24 bottles of beer and a box of washing powder contains 2 kg of powder within a few grammes. However, this does not apply to many other processes: the process data is inexact but to what extent is not known. In these cases information about unreliability can be obtained by comparing different sources, verification with mass and energy balances, comparing elemental analyses, etc.

Sensitivity analysis can also be used to determine the process data which is most important when estimating the reliability of the environmental profile or the inventory table. A method has been developed (Heijungs, 1992) to indicate the magnitude of the change in the result due to a change in the process data. This method is also relevant in the context of the improvement analysis where it will be discussed as *marginal analysis*. This method is also useful for assessing the reliability of the environmental profile as it identifies the process data which the result is most sensitive to. This means that this process data will provide a good starting point for the improvement analysis. The reliability analysis shows that this process data is essential data which therefore has to be determined with great accuracy.

The spread of the data to be used is sometimes also known for the classification factors. The effect of the error on the classification factors is given by

$$\Delta\gamma_l = \sum_{k=1}^s (c_{kl}\Delta\beta_k + \beta_k\Delta c_{kl}) \quad (4.8)$$

Generally speaking the weighting factors used to determine the environmental index will be less accurate than most other data, but the inaccuracy will be unspecified. If they are known (as  $\Delta\mu_l$ ) the following applies to  $\Delta M$ :

$$\Delta M = \sum_{l=1}^m (\mu_l\Delta\gamma_l + \gamma_l\Delta\mu_l) \quad (4.9)$$

#### 4.2.2 Validity analysis

Validity analysis is used to determine how choices and assumptions made affect the results. This is particularly important for debatable choices and assumptions. The reason for this is that even the development of a uniform method should not rule out future discussion.

During each LCA the researcher will have to take some decisions. Some examples will follow, based on a study of the environmental effects of disposable cups (Van Eijk *et al.*, 1991).

When determining the functional unit it could be considered whether one drink of coffee or a fixed quantity (i.e. volume) of coffee should be used. The choice could be varied in the validity analysis to determine the change in the results. If the major conclusions remain unchanged it may be concluded that that study has a high validity in that respect.

The cutlery for stirring was excluded from the analysis when the system boundaries were defined.



As porcelain cups are generally associated with metal spoons and disposable cups with a plastic stirrer the differences could be analysed to provide an indication of the validity of the analysis.

A particular allocation method was selected when allocating the process data. Alternative methods could be studied in the validity analysis.

In addition to these choices made by the researcher an LCA will also include assumptions. The process data will be based on a certain energy model, the number of times returnable packaging is used has to be estimated and a depreciation rate has to be assumed for certain capital goods. The effects of all these assumptions can be studied in the validity analysis.

By contrast with the reliability analysis, a simple rule or protocol cannot be defined for the validity analysis. The choices and assumptions made throughout the life cycle assessment have to be considered.

When two environmental indexes are compared a simple formula can be used to assess the validity of the result. Both the environmental indexes were obtained by aggregating the effect scores (obtained by using the best scientific expertise) with the weighting factors. Weighting factors cannot be determined with the same objectivity as classification factors (the problems associated with the definition of weighting factors were discussed in §4.1). Hence the effects of choosing other weighting factors requires further study.

When two products are considered the following situation arises. Product A has an environmental index  $E_A$

$$E_A = \sum_{l=1}^m \mu_l \gamma_{lA} \quad (4.10)$$

while product B has an environmental index  $E_B$

$$E_B = \sum_{l=1}^m \mu_l \gamma_{lB} \quad (4.11)$$

The difference  $E_{AB} = E_A - E_B$  is examined. If it is positive product B is preferred, if it is negative product A is preferred. If it is zero there is no difference and no preference\*. Hence we are interested in the properties of the expression

$$E_{AB} = \sum_{l=1}^m \mu_l (\gamma_{lA} - \gamma_{lB}) \quad (4.12)$$

It is important to know how sensitive the sign of  $E_{AB}$  is to changes in one or more  $\mu_l$ . Let us say that  $\mu_l$  is changed such that the difference  $E_{AB}$  is exactly 0. The changed situation is indicated by accents  $E_{AB}'$  and  $\mu_l'$ . Hence

$$E_{AB}' = E_{AB} - (\mu_l' - \mu_l) (\gamma_{lA} - \gamma_{lB}) = 0 \quad (4.13)$$

or

$$\mu_l' - \mu_l = \frac{-E_{AB}}{\gamma_{lA} - \gamma_{lB}} \quad (4.14)$$

The relative change in  $\mu_l$  is indicated as  $\varepsilon_l$  and is defined as

$$\varepsilon_l = \frac{\mu_l' - \mu_l}{\mu_l} = \frac{-E_{AB}}{\mu_l (\gamma_{lA} - \gamma_{lB})} \quad (4.15)$$

To ensure that the weighting factors are reasonable a constraint can be imposed on this parameter: the relative change  $\varepsilon_l$  required for each  $l = 1, \dots, m$  where  $\gamma_{lA} \neq \gamma_{lB}$  should be greater than a predefined threshold  $\varepsilon$ . For example,  $\varepsilon$  could be 0.25:

$$\forall i: |\varepsilon_i| > \varepsilon \quad (4.16)$$

\* When comparing  $N$  products  $\frac{1}{2}N(N-1)$  paired comparisons will be made (similar to an unweighted comparison) to rank the product alternatives on an ordinal scale of environmental-friendliness.

Apart from comparison with a threshold specification of the required values  $\epsilon_i$  is also useful.

When faced with difficult choices reversal points can be calculated, similar to the definition of weighting factors which reverse the outcome. The reversal points for the number of times reusable crockery was used were calculated for a study of reusable and disposable crockery (Van Eijk *et al.*, 1991). Such reversal points can be calculated on the basis of the inventory table, the environmental profile or the environmental index. This calculation was made for each effect score in the study by Van Eijk *et al.*. This made it possible to calculate the reversal point (without weighting factors) for each environmental effect at which reusable crockery is more attractive. The mathematics behind this are as follows: let us assume that the  $i^{\text{th}}$  effect score  $\gamma_i$  of a disposable alternative A and an  $N$  times reusable alternative B are considered. When the disposable alternative is used  $N$  times we obtain

$$\gamma_{iA} = Nv_A \quad (4.17)$$

where  $v_A$  represents the contribution to the effect score every time alternative A is used. For the reusable alternative this parameter consists of a fixed part  $f_B$  (due to manufacturing, etc.) and a variable part  $v_B$  which is proportional with the number of times it is used (due to cleaning, etc.):

$$\gamma_{iB} = f_B + Nv_B \quad (4.18)$$

The product alternatives are equivalent if  $\gamma_{iA} = \gamma_{iB}$  or when

$$N = \frac{f_B}{v_A - v_B} \quad (4.19)$$

The number of reuses ( $N$ ) calculated in this way can then be assessed.

During the classification artificial classification factors can be used for substances for which there is no classification factor, to reverse the result. It can then be discussed how reasonable or unreasonable these classification factors are.

### 4.3 Discussion

*La devise de Montaigne était: Que sais-je?  
et la tienne est: Que ne sais-je pas?*  
VOLTAIRE, Dictionnaire philosophique portatif.

We have indicated above how a comprehensive appraisal of one or more products can be based on an environmental profile. Further weighting or aggregation may be needed as in practice there will generally not be one product alternative which does better on all effect scores. An expert committee could be set up to appraise the different effects and to consider qualitative aspects, uncertainties, etc. Such a committee could also be asked to appraise the study to identify unidentified uncertainties, incorrect assumptions or data, errors and unclear language. This would be similar to an environmental impact assessment (EIA). The advantages of this approach will be self-evident: the assessment and verification will be unbiased.

However, the work required is the main disadvantage of such a procedure. An EIA concerns a limited number\* of major planning decisions. The procedure to arrive at a decision takes several months and involves many people. However, product assessments have to be made for thousands of products every year, even without including prototypes and minor improvements in the design†. It would not be feasible for all of them to be assessed by an expert committee. One of the aims of this project is to contribute to the assessment method. This should reduce the uncertainties about products which are preferable in environmental terms (Guinée *et al.*, 1992; Mieras, 1992). The institution of a code of conduct could also contribute to this process. Thirdly an appeal committee or a peer reviewer panel could give its comments in exceptional cases, either when invited or on its own

\* The 1990 annual report of the Netherlands Environmental Impact Assessment Committee lists 100 reports.

† A fairly large supermarket carries a range of 8000 articles. Assuming that many manufacturers introduce new versions or packaging every other year even a shop like this would involve 4000 LCAs annually.

initiative.

Clearly such strict requirements need not be applied to applications within a company. Rough estimates are particularly useful when designing or redesigning products. The whole range of requirements concerning reliability, clarity and reproducibility only become important when an LCA is used for promotion purposes. There is a particular need for an approximate rating of environmental effects for the large number of in-company applications.

The need for a formal method will increase if it is decided to develop a method for environmental life cycle assessments of products, to institute a code of conduct for their implementation and reporting and to set up an arbitration committee. As referred to above this will require the definition of evaluation factors, similar to the classification factors.

A report on the latest developments, considerations and data could be published regularly once a widely accepted procedure for the definition of such evaluation factors has been set up, based either on the natural sciences or on the social sciences. This would be similar to the classification where bodies such as the *Intergovernmental Panel on Climate Change* (IPCC) guarantee the coordination and integral publication of the latest information about the greenhouse effect and the most recent GWPs.

The method described in §4.1 provides a basis for the implementation of the concept *environmental capacity*\*. If levels of sustainability can be determined on a scientifically acceptable basis each effect score of a product will be associated with a parameter with the dimension time. This time can be interpreted as the period during which the Earth is exposed to the environmental problems under consideration (see footnote † on page 109) up to the limits set. If the problem associated with the weighting factor  $\psi_i$  can be solved, then the parameters can be added. The sum of these parameters is the time for all the problems combined. To distinguish this time unit from normal seconds it could be called the *environmental second*. The environmental index of a product expressed in environmental seconds, multiplied by the number of products consumed throughout the world in a year results in a dimensionless parameter. If this parameter was calculated for all products on earth and totalled an expression would be obtained to indicate the extent to which environmental capacity is used or exceeded. The "environmental capacity" is exceeded when more than one environmental second is "consumed" per second of normal time. If this is the case environmental seconds are being borrowed and as borrowing means making a claim on the future we are reminded that we should not steal our childrens' birthright.

\* This argument is similar to that of De Groot (1992) in which an activity is normalised by life span to eliminate time from the expression. The sustainability criterion used in this context is based on the area of solar panels required to remedy the problem.



## CHAPTER 5

# IMPROVEMENT ANALYSIS

The status of the fifth component of an environmental life cycle assessment is slightly different from that of the other components. An improvement analysis is only carried out if product innovation is the aim of the study. Normally an improvement analysis will not be undertaken if the study is used to compare products. The initial four components, i.e. goal definition, inventory analysis, classification and evaluation, are undertaken in that order. As the inventory analysis produces a result which can be used independently of the other components a life cycle assessment could be completed after the inventory analysis. The same applies to the classification, the environmental profile can be used on its own. However, this is not likely to become widespread, in most cases a further rating will have to be made in the classification and the evaluation. For product innovation it may be attractive to omit the classification and the evaluation and progress directly from the inventory analysis to the improvement analysis\*. The classification also provides an excellent start for the improvement analysis, without an evaluation. The special nature of the improvement analysis is clearly illustrated in the figure showing the general structure (Figure 0.2 on page 5).

During an improvement analysis information from the process tree is used to make recommendations for the redesign of a product or changes in the process operations. The improvement analysis results in well-defined suggestions for the redesign of products or processes. A number of product variations will be designed after a comprehensive assessment of the technical and financial feasibility and other aspects of these suggestions. These designs will then be assessed by comparing them with each other as well as with the original product, in a comparative LCA. You are referred to §1.1 and Figure 1.1 for further information about the method for product innovation.

Information has been lost due to the aggregation of environmental interventions throughout the process tree. As a result of this the environmental interventions are now easier to comprehend. During the improvement analysis the information which was lost is now used to identify the places where changes can be made on the basis of the details of the process tree (processes and interventions).

Two analytical techniques are distinguished in the guide:

- dominance analysis (step 5.1);
- marginal analysis (step 5.2).

Both techniques can be used to make recommendations about these starting points. They will be discussed in the following two sections below.

\* In a way this also leads to a classification or evaluation: when the improvement analysis follows the inventory analysis the aspect to be improved has to be selected, e.g. would we rather reduce Hg or SO<sub>2</sub> emissions? Such an evaluation need not be complete, Figure 1.1 refers to a limited evaluation of this kind.

## 5.1 Dominance analysis

During the fourth step of the inventory analysis the extent to which each process in the process tree occurs is determined. The occurrence of each process is illustrated by the matrix method described in §2.4. This information was used to calculate the inventory table and the environmental profile and possibly the environmental index too. The long description was reduced to a manageable list by aggregation throughout the entire product system. This will generally be sufficient for a comparative assessment.

However, information which may be relevant to the improvement analysis has been lost by the aggregation. To analyse the causes of the environmental interventions associated with a product system information is required about the occurrence of each process. Dominance analysis provides such a causal analysis. A method of presenting the process data will be described below. When dominance analysis is used the investigator has to find a path through the network of causal relationships revealed by this presentation\*. The objective of this analysis is to provide information about the extent to which environmental interventions or effects can be attributed to (possibly variable) properties of the product or processes. Is the acidification associated with milk bottles largely due to the aluminium cap, rinsing, glass production or transport? The quantified and allocated individual process data provide the foundations for such analyses.

The processes which are responsible for each part of every environmental intervention are determined. The process emission responsible for each part of every effect score or environmental index is also determined. Thus this method is used to identify the dominant elements in the inventory table, the environmental profile and the environmental index. These three levels will be discussed below.

A product's inventory table is created by aggregation of the interventions of the individual processes. The dominant elements can be traced by analysing the individual contributions of each process. The mathematical expression of this is relatively straightforward.

The  $k^{\text{th}}$  environmental intervention is obtained by multiplying the occurrence of the  $i^{\text{th}}$  process  $p_i$  by the relevant intervention per unit of that process  $b_{ki}$  and by aggregating this across all processes:

$$\beta_k = \sum_{i=1}^q b_{ki} p_i \quad (5.1)$$

Detailed information about the processes is lost at this stage. Information about the interventions of a process by ratio of the occurrence of that process is obtained by carrying out the multiplication, but not the aggregation, and by placing the numbers obtained in a matrix. This illustrates the contribution each process makes to each environmental intervention. The matrix containing these numbers can be extended with an additional column containing the aggregated inventory table. The *process matrix* is defined by doing this for the economic part as well as the environmental part. The process matrix  $P$  has a dimension  $(r+s) \times (q+1)$  and is represented by

\* In a sense dominance analysis is not a method but rather a notation method which can be analysed.





5.1 Dominance analysis

During the fourth step of the investigation each process in the process tree occurs is determined. The occurrence of each process is determined by the matrix method described in §2.4. This information was used to reduce the environmental index to a manageable list by aggregation throughout the entire process tree. This information was used to reduce the environmental index to a manageable list by aggregation throughout the entire process tree.

$$\begin{pmatrix}
 a_{11}P_1 & \dots & a_{1i}P_i & \dots & a_{1q}P_q & \alpha_1 \\
 \dots & \dots & \dots & \dots & \dots & \dots \\
 a_{j1}P_1 & \dots & a_{ji}P_i & \dots & a_{jq}P_q & \alpha_j \\
 \dots & \dots & \dots & \dots & \dots & \dots \\
 a_{r1}P_1 & \dots & a_{ri}P_i & \dots & a_{rq}P_q & \alpha_r \\
 \gamma_{11} & \dots & \gamma_{1i} & \dots & \gamma_{1q} & \gamma_1 \\
 \dots & \dots & \dots & \dots & \dots & \dots \\
 \gamma_{i1} & \dots & \gamma_{ii} & \dots & \gamma_{iq} & \gamma_i \\
 \dots & \dots & \dots & \dots & \dots & \dots \\
 \gamma_{m1} & \dots & \gamma_{mi} & \dots & \gamma_{mq} & \gamma_m
 \end{pmatrix} \tag{5.6}$$

where the  $i^{th}$  effect score of the  $i^{th}$  process  $\gamma_{ii}$  is represented by

$$\gamma_{ii} = \sum_{k=1}^s c_k b_{ki} P_i \tag{5.7}$$

The dimension of this process matrix is  $(r+m) \times (q+1)$ , normally  $m \ll s$ . This method corresponds with that described earlier, the only difference is that dominant elements are traced at the effect level rather than at the intervention level. In the example of the aluminium packaging it will be found that the packaging produces the major element of acidification.

Once the environmental index has been compiled on the basis of the weighting factors the dominance analysis can be carried out at the evaluation level. In this way dominant elements in environmental pollution can be determined. For example, the fact that 30% of the environmental index is caused by the aluminium packaging will be valuable information in the improvement analysis. The process matrix takes the form

$$\begin{pmatrix}
 a_{11}P_1 & \dots & a_{1i}P_i & \dots & a_{1q}P_q & \alpha_1 \\
 \dots & \dots & \dots & \dots & \dots & \dots \\
 a_{j1}P_1 & \dots & a_{ji}P_i & \dots & a_{jq}P_q & \alpha_j \\
 \dots & \dots & \dots & \dots & \dots & \dots \\
 a_{r1}P_1 & \dots & a_{ri}P_i & \dots & a_{rq}P_q & \alpha_r \\
 M_1 & \dots & M_i & \dots & M_q & M
 \end{pmatrix} \tag{5.8}$$

where the environmental index of the  $i^{th}$  process is given by

$$M_i = \sum_{l=1}^m \sum_{k=1}^s \mu_l c_k b_{ki} P_i \tag{5.9}$$

The causal relationships to be traced by a dominance analysis may be extremely complex\*. Marginal analysis is a method which can reveal these relationships in a much more direct manner. Furthermore it uses the dynamic properties of the process tree. Marginal analysis is discussed below.

\* An interesting question arises: can the search for major axes (= dominant processes?) be made easier by singular value decomposition?

## 5.2 Marginal analysis

In dominance analysis the entire process tree has to be traced to find the ultimate cause of a given dominant element in the inventory table, environmental profile or environmental index. However, such dominant elements may be due to many causes. In many processes fossil energy sources are converted to energy, which is accompanied by emissions of  $\text{SO}_2$ , for example, which leads to acidification. Generally it will be found that energy generation is a dominant element in  $\text{SO}_2$  emissions and acidification, either directly or indirectly. The energy is supplied to a wide range of processes which are associated with the product system in any of a number of ways.

When improving products it is important to know which *changes* in the process operation will have the desired effect. Changes in process operations which will produce this effect in the most *efficient* way are particularly important, as the effects depend on many processes. Hence, we are interested in the change in environmental burden due to a change in the process data. One way of doing this might be to change all the process data in an LCA, one item after another, and to determine the percentage change in a given environmental intervention, effect score or environmental index. The process data in which a small change will lead to a major improvement are the most important for product improvement. The feasibility of such a small change is has to be assessed separately. By implementing the mathematical method described below in software the most efficient options for product improvement can be found by "just a press of the button". A designer or process engineer with practical experience of the process can then select the most useful suggestions on that list.

This operation would be extremely tedious if it were carried out manually as the entire process tree has to be calculated whenever a change is made. This could be done by a computer program. However a more suitable method can be developed with special methods for calculating the process tree (§2.4). This method is known as *marginal analysis*. The essence of this method is that apart from calculating incidences of each process a tangent is drawn which indicates the extent to which the incidence of the process will change after a marginal change in the process data\*. In this way the dynamic properties of the process tree are included. This does not require any additional information: in principle all information about the functional links between process data and interventions, effects and index is known but most of this information is not used to create the inventory table, etc. Marginal analysis employs some of this information which would otherwise remain hidden, by using the magnitude and the tangent. This is illustrated in Figure 5.1.

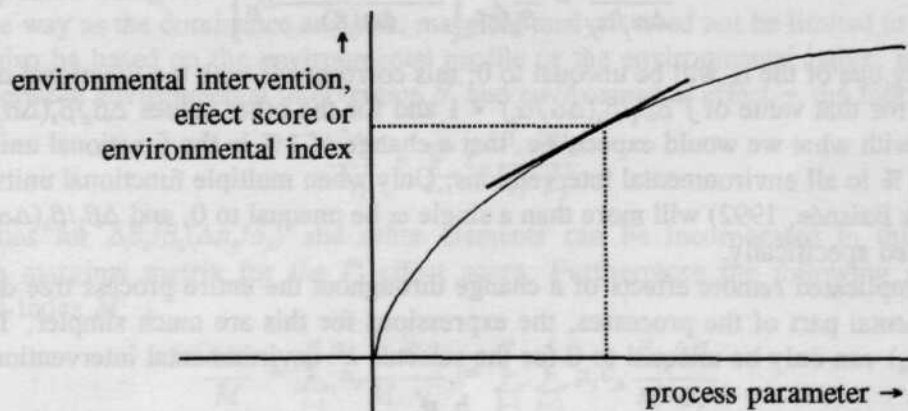


FIGURE 5.1. In marginal analysis the magnitude and the *tangent* of the environmental intervention, effect score or environmental index is determined when one of these is plotted as a function of a process parameter.

Marginal analysis starts on the basis of the environmental index or a single effect score or

\* This tangent is only valid for marginal changes because the relationship is non-linear.

environmental intervention. A *marginal matrix* can be constructed for the selected parameter. This matrix represents the effect of a marginal change in each process parameter (i.e. for all processes in the process tree and for economic parameters as well as environmental intervention parameters) for the selected parameter. For example: we are interested in the change in a selected environmental intervention  $\beta_k$  due to a change in process parameter  $a_{ji}$ , in other words, in  $\Delta\beta_k$  as a function of  $\Delta a_{ji}$ , represented by  $\Delta\beta_k(\Delta a_{ji})$ . This becomes clearer by considering the relative change, where  $\Delta\beta_k/\beta_k(\Delta a_{ji}/a_{ji})$  indicates the amplification factor of a percentage change.

The mathematical derivation of the marginal matrix is based on differential calculus and matrix algebra which will not be repeated here. For further information you are referred to Heijungs (1992).

At the inventory analysis level the marginal matrix for the  $k^{\text{th}}$  intervention is

$$D_k = \begin{pmatrix} \left[ \frac{\Delta\beta_k/\beta_k}{\Delta a_{ji}/a_{ji}} \right]_{\substack{i=1,\dots,q \\ j=1,\dots,r}} & \left[ \frac{\Delta\beta_k/\beta_k}{\Delta\alpha_j/\alpha_j} \right]_{j=1,\dots,r} \\ \left[ \frac{\Delta\beta_k/\beta_k}{\Delta b_{ni}/b_{ni}} \right]_{\substack{i=1,\dots,q \\ n=1,\dots,s}} & \left[ \frac{\Delta\beta_k/\beta_k}{\Delta\beta_n/\beta_n} \right]_{n=1,\dots,s} \end{pmatrix} \quad (5.10)$$

The upper left section is a matrix with a dimension  $r \times q$  containing the marginal coefficients for a change in the economic flows. As a change in a single economic flow of one process will influence the incidence of at least one other process the influence of the marginal change on the selected parameter is rather complex. The following applies

$$\frac{\Delta\beta_k/\beta_k}{\Delta a_{ji}/a_{ji}} = -a_{ji} \frac{(-1)^{j+i} \det(A_{ji})}{\det(A)} + \frac{a_{ji}}{\beta_k} \frac{(-1)^{j+i}}{\det(A)} \sum_{\substack{l=1 \\ l \neq i}}^q (b_{li} \det(A^l_{ji})) \quad (5.11)$$

where  $A_{ji}$  represents matrix  $A$  in which the  $j^{\text{th}}$  row and  $i^{\text{th}}$  column have been removed (the *minor*; see e.g. Apostol, 1969).  $A^l_{ji}$  is the minor of matrix  $A^l$  ( $A$  in which the  $i^{\text{th}}$  column has been replaced by the kernel process  $\alpha$ ).

The following applies to a change in one of the  $\alpha_j$  (in the upper right hand corner of the marginal matrix):

$$\frac{\Delta\beta_k/\beta_k}{\Delta\alpha_j/\alpha_j} = \frac{\alpha_j}{\beta_k} \sum_{i=1}^q \left[ \frac{(-1)^{j+i} \det(A_{ji})}{\det(A)} b_{ki} \right] \quad (5.12)$$

Normally only one of the  $\alpha_j$  will be unequal to 0; this corresponds with the functional unit (see §2.4). In this event for that value of  $j$   $\Delta\beta_k/\beta_k(\Delta\alpha_j/\alpha_j) = 1$  and for the other values  $\Delta\beta_k/\beta_k(\Delta\alpha_j/\alpha_j) = 0$ . This corresponds with what we would expect, i.e. that a change of 1% in the functional unit will result in a change of 1% in all environmental interventions. Only when multiple functional units are used (see e.g. Heintz & Baisnée, 1992) will more than a single  $\alpha_j$  be unequal to 0, and  $\Delta\beta_k/\beta_k(\Delta\alpha_j/\alpha_j)$  will have to be calculated specifically.

As the complicated remote effects of a change throughout the entire process tree do not occur in the environmental part of the processes, the expressions for this are much simpler. The expression  $\Delta\beta_k/\beta_k(\Delta b_{ni}/b_{ni})$  can only be unequal to 0 for the selected  $k^{\text{th}}$  environmental intervention. Hence:

$$\frac{\Delta\beta_k/\beta_k}{\Delta b_{ni}/b_{ni}} = \begin{cases} \frac{b_{ki} p_i}{\beta_k} & (\text{if } n=k) \\ 0 & (\text{otherwise}) \end{cases} \quad (5.13)$$

The section in the right hand corner was only included to obtain symmetry, its meaning is irrelevant:

$$\frac{\Delta\beta_k/\beta_k}{\Delta\beta_n/\beta_n} = \begin{cases} 1 & (\text{if } n=k) \\ 0 & (\text{otherwise}) \end{cases} \quad (5.14)$$

It is relatively easy to calculate the marginal matrix for each desired environmental intervention by



using a computer program to calculate the incidence of each process using the matrix method\*. This will produce a screen full of numbers, many of which will be 0. The next step is to find the relevant information in the matrix. This is done by making a list in which the elements in the marginal matrix are arranged in order of their absolute magnitude. The elements in a given process, economic flow or environmental intervention where a small change will have the largest effect on the selected environmental intervention will be at the top of the list. Hence these are the options offering the best prospects for product improvement. Some elements in the marginal matrix may even exceed 1. This type of process data is particularly interesting as it provides the opportunity to bring about a major change in the inventory table through a minor change in the process operations.

Not all suggestions for product improvement suggested by marginal analysis will be useful. The analysis might suggest an improvement in the efficiency of the production processes, e.g. by manufacturing more than 1 kg of rolled steel from 1 kg steel. Such impossibilities will be clear even to outsiders, but this may be more difficult for many processes. Hence an understanding of process engineering is essential during the improvement analysis to distil useful suggestions from the many leads given by marginal analysis.

Marginal analysis serves another function, as referred to above during the discussion of the sensitivity analysis (§4.2). This method not only identified the process data which hold the best prospects for improvement, it also picks out the process data where inaccuracies will have the greatest consequences. Hence, not only should the usefulness of the suggestions from the marginal analysis be considered, it is also necessary to investigate the accuracy of the data.

Marginal analysis provides information about individual changes in all process data provided that all other process data remains constant. This is unrealistic in practice. When the marginal analysis suggests that soot emissions should be reduced this means that the process operations have to be changed which will also affect other process data: the process conditions, such as the combustion temperature, will have to be changed or a flue gas scrubber will have to be installed, in which case another process is added. Hence, the suggestions provided by the marginal matrix which result in a revision of the design of the process or product will have to be compared to the original design in a separate LCA (see Figure 1.1). The result may be better or worse: a change in combustion temperature will affect the use of resources and the creation of emissions and residues. In this way the main premise of life cycle assessment - that shifts towards other processes or effects have to be avoided - is observed. Another reason for examining the improvement is that the non-linear effect of a change may be different from the prediction based on the marginal matrix as the matrix is only valid for infinitesimally small changes.

In the same way as the dominance analysis, marginal analysis need not be limited to the inventory table, it can also be based on the environmental profile or the environmental index. Because of the relationship between environmental intervention  $\beta_k$  and environmental effect  $\gamma_l$  the following applies:

$$\frac{\Delta\gamma_l}{\gamma_l} = \sum_{k=1}^s c_{lk} \frac{\beta_k}{\gamma_l} \frac{\Delta\beta_k}{\beta_k} \quad (5.15)$$

The expressions for  $\Delta\beta_k/\beta_k$  ( $\Delta a_{ji}/a_{ji}$ ) and other elements can be incorporated in this equation to determine the marginal matrix for the  $l^{\text{th}}$  effect score. Furthermore the following applies to the environmental index  $M$ :

$$\frac{\Delta M}{M} = \sum_{l=1}^m \mu_l \frac{\gamma_l}{M} \frac{\Delta\gamma_l}{\gamma_l} = \sum_{l=1}^m \sum_{k=1}^s \mu_l c_{lk} \frac{\beta_k}{M} \frac{\Delta\beta_k}{\beta_k} \quad (5.16)$$

Clearly there are advantages associated with undertaking an improvement analysis of a product system on the basis of a given theme or at an even higher level.

\* Most of the calculations are required to calculate the section in the upper left hand corner. For all  $a_{ji}$  which equal 0 the corresponding element of  $D_k$  will also be 0; the time required for the calculation can also be reduced by using the datum that many  $b_{ki}$  are also 0, to limit the summation.

5.3 Discussion

*Que la nature est corrompue. Par la nature même.  
Qu'il y a un réparateur. Par l'Écriture.*  
BLAISE PASCAL, *Pensées.*

The improvement analysis introduces a method for directed intervention in human activities. In principle, there is no limitation to the types of activities which can be investigated with the aid of an LCA. An LCA can be used as an instrument for assessment as well as as an analytical instrument to reduce the harmful effects of these activities. The development of improvement analysis is still in its infancy. As a result the strength or weakness of these analytical techniques has yet to be proven. It is likely that other methods could be used. In future it may even be possible to develop an efficient protocol for improvement analysis.

This discussion again underlines the need for computer programs to carry out LCAs. A basic process tree can be calculated without the aid of a sophisticated program, but the calculation of the marginal matrix and variations (which are also needed for the reliability analysis) requires extensive calculations. As the formulas to be implemented are given in this report the development of such a program only requires user-friendly programming of all the options required and the availability of a reliable file of basic process data. The availability of such a file remains a matter of concern.

Once more we would like to stress the need to compare a supposedly improved product with the original design. *Our Common Future* (WCED, 1988) contains the following section about this aspect: "Technology will continue to change the social, cultural and economic fabric of nations and the world community. With careful management, new and emerging technologies offer enormous opportunities for raising productivity and living standards, for improving health, and for conserving the natural resource base. Many will also bring new hazards, requiring an improved capacity for risk assessment and risk management." The iterative use of life cycle assessment to improve products appears to be particularly suitable for this as the method is essentially based on a consideration of the entire chain and all environmental effects.

$$\frac{\Delta \delta / \delta}{\Delta \delta / \delta} = \begin{cases} 1 & (i = j) \\ 0 & (\text{otherwise}) \end{cases} \quad (5.14)$$

**APPENDIX A**

**CONTRIBUTORS**

A large number of people contributed to this study and its coordination. It would be impossible to list everyone who contributed information on whatever scale. Hence we will limit ourselves to listing the members of the steering committee and the think-tank as well as those people to whom special thanks are due.

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We would also like to acknowledge the support of the following:

J.A. Assies — CML  
 Th. Baumgartner — IÖW  
 H. Blonk — B&G  
 J. Broers — Rijkswaterstaat  
 M. Elbers — CML  
 G. Finnveden — IVL  
 R. Frischknecht — ETH  
 P. Hofstetter — ETH  
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 I. Knoepfel — ETH  
 F. de Leeuw — RIVM  
 P.A.A. Mulder — CML  
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## APPENDIX B

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