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Development of a methodology for the environmental life-cycle assessment of products: with a case study on margarines

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DEVELOPMENT OF A METHODOLOGY FOR THE ENVIRONMENTAL LIFE-CYCLE ASSESSMENT OF PRODUCTS

with a case study on margarines

ONTWIKKELING VAN EEN METHODE VOOR DE MILIEUGERICHTE LEVENSCYCLUSANALYSE VAN PRODUKTEN

met een voorbeeldtoepassing op margarines

PROEFSCHRIFT

TER VERKRIJGING VAN DE GRAAD VAN DOCTOR
AAN DE RIJKSUNIVERSITEIT TE LEIDEN,
OP GEZAG VAN DE RECTOR MAGNIFICUS DR L. LEERTOUWER,
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Voor mijn vader

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PREFACE

This thesis has been written based on research carried out at the Centre of Environmental Science of Leiden University (CML) during the period 1989 to 1994, particularly in the section Substances & Products. Parts of this thesis are based on joint research within this section and in fact all members of this section have made their contributions to this thesis, for which I am very grateful. Furthermore, I am grateful for the contributions and support by CML- and other colleagues.

Part of the research has been done within the framework of the study *Towards a method for comparative product assessment on environmental effects*. This study was carried out as part of the Netherlands National Reuse of Waste Research Programme (NOH) which is funded by the Netherlands Ministry of Economic Affairs and the Directorate-General for Environmental Management of the Ministry of Housing, Planning and Environment. A database put at my disposal by the company Van den Bergh Nederland formed the basis of the case study described in this thesis.

Chapter 1

Introduction

1.1 DEFINITION AND SCOPE OF LCA

Environmental life-cycle assessment (LCA) is a tool for assessing the environmental impacts of a product, or more precisely, of a product system required for a particular unit of function. The term "product system" is taken to mean the product throughout its entire life-cycle, from cradle to grave, in terms of all the economic processes involved. The term economic process - employed as the converse of environmental process - refers to any kind of process producing an economically valuable material, component or product, or providing an economically valuable service such as transport or waste management. LCA takes as its starting point the function fulfilled by a product system and, in principle, takes into account as far as possible all the environmental impacts of all the processes needed to fulfil this function - from resource extraction, through materials production and processing and consumption of the product, to waste processing of the disposed product. LCA methodology forms the general topic of this doctoral thesis.

LCA as defined in this thesis deals only with the environmental impacts of a product and does not include other aspects such as financial, technical and macro-political aspects (e.g. third world issues). This does not of course imply that these other aspects are less relevant for the overall evaluation of a product. Thus, Osnowski & Rubik [1] and Pedersen Weidema [2] suggest that LCA might be seen as part of a more comprehensive life-cycle assessment of products that also includes safety aspects, economic aspects and social aspects such as employment, unequal wages and working conditions. In a study by McKinsey & Company [3] and a study by de Wit *et al.* [4] separate life-cycle assessments within a joint framework were performed for both (micro-)economic and environmental aspects. Because in this thesis the scope is limited to environmental aspects, the term LCA is used in the sense of environmental LCAs.

LCA is a decision support tool, and not the decision itself. LCAs generate information that can be used in decision-making by governments, businesses and consumers. To further define the scope and range of LCA, it is useful to compare this tool with other environmental decision support tools such as Substance Flow Analysis (SFA) [cf. 5] (also known as material balances), Technology Assessment (TA), Environmental Impact Assessment (EIA), Risk Assessment (RA) and Environmental Audit (EA). As discussed by Udo de Haes and Huppes [6] and Heijungs *et al.* [7], all these tools have different prime economic objects of analysis. LCA analyses the environmental impacts of a product through its entire life-cycle; SFA analyses the flows and accumulations of one substance (or substance group) through the entire life-cycle of that substance (group) within a defined region; TA assesses environmental, social, economic and other relevant aspects of future technologies; EIA analyses the environmental impacts of investments and plans envisaged for specific locations; RA analyses the adverse impacts of technical plant; a distinction can be made between RA in a strict sense and RA in a broader sense; the former analyses very small probabilities of extremely adverse effects from one plant in a specific location, while the latter considers risks to be any adverse effects of a plant occurring with a certain probability; EA, finally, deals mainly with the environmental performance of individual business units or firms [6]. These tools also differ in the degree to which an entire production chain (life-cycle) is considered in the analysis: apart from LCA, the life-cycle approach is found mainly in SFA and, to a lesser degree, in TA.

The different scopes of the various tools also have important implications for the

methodology used for impact assessment. This is due to the different time and space characteristics of the various tools. For example, RA in a broader sense allows statements to be made on toxicity in terms of actual risks, e.g. concentrations exceeding a particular threshold value, because RA focuses on processes at one specific site. With LCA, however, only *potential* impacts can be assessed, one reason being that the time dimension is not taken into account in process emission data. This problematical issue is illustrated by an example in the epilogue of Chapter 4 of this thesis. Because of these differences, the tools mentioned each have a specific role to fulfil and to a large extent yield complementary information.

The concept of LCA is applicable not only to products but also to materials [cf. 8,9]. However, an LCA of a material usually goes no further than production of the material, excluding further processing in products and waste processing of the material. This type of LCA is therefore often referred to as cradle-to-gate LCA.

Under the umbrella of LCA there is still quite a wide range of methodological approaches possible. Two distinctions are made here to further specify the scope of this thesis: qualitative LCAs based on (partly) non-exclusive criteria versus quantitative LCAs based on (as far as possible) exclusive criteria; and "steady-state" LCAs versus "dynamic" LCAs.

LCAs, as used today, vary from more qualitative methods using non-exclusive criteria to predominantly quantitative methods using exclusive criteria. Both types of method are designed to assess the environmental impacts of the entire life-cycle of a product [22]. Examples of a predominantly qualitative approach based on non-exclusive criteria are the PVC study of Christiansen *et al.* [10], the study of Fraanje *et al.* on the environmental impact of house-building [11], the German "Produktlinienanalyse" approach [1], and the approach described by Van Weenen [12] focusing on product design applications. The criteria employed in these approaches include, for example, recyclability, repairability, life-span, types of resources used, and types of substances emitted. Some of these criteria may well be of a quantitative nature. Thus, if in a qualitative approach criteria are mapped on an ordinal scale in a scoring table for all phases of the life-cycle, for example, the different classes of the ordinal scale may be distinguished by criteria of a cardinal quantitative nature (see, for example, the study of Fraanje *et al.* [11]). Quantitative approaches, on the other hand, aim to quantify the environmental impacts of all the constituent processes of a product's life-cycle on a cardinal scale with respect to a number of exclusive criteria [13,14,15,16,17]. They may also include qualitative information about the processes involved. In such approaches, recyclability, repairability and life-span are not separate criteria, but are included in the (cardinally quantified) resources required and substances emitted by the product system studied [18]. This thesis aims to present a quantitative approach based on an exclusive set of criteria. Little attention has been paid to qualitative aspects, although it is recognized that qualitative information on environmental aspects that are unquantifiable is a useful supplement to a basically quantitative approach.

Environmental impacts during the life-cycle of a product occur at different moments in time. For example, the environmental impacts of the capital goods and materials needed for the functioning of a product today have taken place in the past and the environmental impacts of the waste management of the product will take place in the future. This time aspect of LCA can be resolved in two different ways: 1) by generally assuming all processes to take place at the same moment in time - thus neglecting old technologies and assessing processes according to the

(average) modern or projected technology, as appropriate to the goal of the study - and adding the associated environmental impact without specifying or allowing for the time in any way [cf. 4]; or 2) by quantifying and adding the environmental impacts of processes specified in time, permitting an analysis to be made of trends in the environmental impact of a product function with time. Moll [19,20,21] terms the first type a "steady-state" LCA and the second type a "dynamic" LCA. According to Moll's terminology, this thesis deals with "steady-state" LCA. However, the majority of methods proposed in this thesis can also be applied to "dynamic" LCA.

1.2 BRIEF HISTORICAL REVIEW

The first studies that are now recognized as (partial) LCAs date from the late sixties and early seventies, a period in which environmental issues like resource and energy efficiency, pollution control and solid waste became issues of broad public concern [22]. The scope of energy analyses [23,24,25], which had been conducted for several years, was later broadened to encompass resource requirements, emission loadings and waste production. One of the first studies quantifying the resource requirements, emission loadings and waste arisings of different beverage containers was conducted by Midwest Research Institute for the Coca Cola Company in 1969. However, it was never published. A follow-up to this study on beverage containers, conducted by the same institute for the U.S. Environmental Protection Agency in 1974, marked the beginning of the development of LCA as we know it today [13]. The Midwest Research Institute used the term Resource and Environmental Profile Analysis (REPA) for this kind of study, which was based on a systems analysis of the production chain of the investigated products "from cradle to grave". After a period of diminishing public interest in LCA and a number of unpublished studies, from the early eighties on there has been rapidly growing interest in the subject. In 1984 the Swiss Federal Laboratories for Materials Testing and Research (EMPA) published an important report [14] that presented a comprehensive list of the data needed for LCA studies, thus catalyzing a broader application of LCA [22]. The study also introduced a method for dividing airborne and waterborne emissions by semi-political standards for those emissions and aggregating them, respectively, into so-called "critical volumes" of air and "critical volumes" of water. In the Netherlands the critical volumes approach was simultaneously and independently developed by Druijff [26]*.

The nineties has seen remarkable growth of LCA activities in Europe as well as in North America, which is reflected in the number of workshops and other forums that have been organized this decade (see Appendix 1, p.15) [27,28,29,30,31,32]. This flourish of activity is also reflected in the number of methodology projects carried out - in the USA [33], Canada [34] and Europe [35,36,37,38] - and Ph.D. theses published [2,20,39,40].

Through its North American and European branches, the Society of Environmental Toxicology and Chemistry (SETAC) plays a leading role in bringing LCA practitioners, users and methodology developers together to collaborate on the continuous improvement of LCA methodology. The SETAC workshop reports [28,30,31] illustrate the methodological and terminological

* For a more comprehensive historical review, see Assies [22].

developments that have occurred during the nineties [41]. In addition, the International Standards Organization (ISO) has established a technical committee (TC 207) concerned with standardization of a number of environmental management tools, including LCA. While SETAC is a private organization that provides a discussion platform for scientists, ISO is a much more formal body which focuses on the development of non-binding agreements among countries.

Before SETAC's adoption of a coordinating role in the field of LCA, LCA covered (and to a less extent still covers) a wide range of (non-transparent) methodologies with numerous differences [cf. 42], which frequently gave rise to a confusion of tongues among LCA practitioners. This resulted in case studies on similar products yielding conflicting results. One of the first milestones to be achieved under the auspices of SETAC to improve this situation is the methodological framework presented in a first "Code of Practice" [43], drafted by an international committee of LCA-experts. In this framework four main components are distinguished: goal definition and scoping, inventory analysis, impact assessment and improvement assessment. Besides this framework, the "Code of Practice" provides definitions for a number of terms and an initial review of possible methodologies for the different components and steps. However, additional efforts are required to facilitate a common LCA language and enable comparison of LCA results [38,44,45]: methods for the various components and steps need to be (further) developed or improved; the availability and quality of the data required for the various components and steps need to be improved; and software should be developed to support practitioners in carrying out LCAs and minimizing the costs of LCA-studies, thus enabling a broader application of LCA [46]. SETAC also intends to coordinate activities in these fields.

1.3 SCIENTIFIC BACKGROUND

LCA is an example of interdisciplinary environmental science. It involves several natural as well as socio-economic sciences, integrating them into a single framework. The aforementioned methodological components differ mainly in the type of knowledge required for each. Goal definition is concerned with information from the different stakeholders in the study, on which there must be agreement in order to specify the terms of reference of the study. The inventory is pre-eminently concerned with systems analysis and process technology. At its core is an analysis of economic systems based on technical process knowledge and on a systems analysis of the inputs from and outputs to some well defined environment, *viz.* the natural environment, due to a set of operations (system) [cf. 47 and 48]. The classification/characterization component is based on the environmental sciences, with the environmental impacts of economic systems being assessed by applying knowledge generated in the various fields of natural science. The valuation component is a subject of the social sciences and is concerned with the values and preferences of society. Common social-science techniques such as multi-criteria analysis - aiming to rank alternatives according to a number of criteria based on an review of the impacts of the alternatives on these criteria [49] - may be applied in this component of LCA. Improvement assessment is based on applied mathematics and process technology. After thorough "scientific shopping", i.e. gathering of relevant knowledge generated by the individual scientific disciplines, an additional scientific and societal value is added by adapting this knowledge to and integrating it in the LCA framework.

1.4 APPLICATIONS

The main categories of LCA application are product improvement, new product design, product information, ecolabelling and the exclusion or admission of products from or to the market. LCA can also be applied to assess policy strategies on matters such as waste management [50,51]. Public attention has focused primarily on product information comparing functionally equivalent consumer products, particularly types of packaging (milk packaging and beverage containers, for example). Today, attention is also being given to the improvement and (re)design of all kinds of products including chairs, carpets, batteries, trains, insulation materials, etc. LCA has good potential for becoming an important support tool for all kinds of environmental product-oriented decisions by government (see, for instance, the Netherlands National Environmental Policy Plan(s) [52,53]), companies and consumers. However, due care should be taken not to have too high expectations, as further work on methodology and data issues is still required (see above).

1.5 GOAL OF THIS THESIS

Work on this thesis was started in September 1989, just before the remarkable revival of LCA. This thesis has been completed in close interaction with the SETAC activities mentioned above. On the one hand, SETAC has provided a platform for presenting and discussing the subject matter of this thesis, and on the other hand this subject matter has constituted a useful input for SETAC discussions during these years.

The main focus of the thesis is development of LCA methodology. It is stressed here that this does not imply that the aforementioned aspects - improvement of data quality and availability and software development - are of minor significance. On the contrary, all these elements are necessary if LCA is to be made to work. A theoretical method with no data is useless and a (quantitative) theoretical method published in lengthy books without any supportive software tools will not be applied in practice. With respect to software development I have done some work outside the framework of this thesis [54,55,56], which has been continued by others and is still being developed.

The purpose of this thesis has been to increase the transparency of LCA as a decision support tool and to improve the scientific basis of LCA, with the greatest focus on the impact assessment component. Improvement of the scientific basis of LCA is understood to mean adaptation of the state of the art of the scientific disciplines on which LCA is based to the aims and needs of LCA. Improving the scientific basis of LCA thus does not mean that new scientific theories are posited or empirical experiments performed, but that a method is designed based - as far as possible - on the present status of relevant scientific disciplines (see above).

A significant part of the work has been done within the framework of the LCA methodology project undertaken in the Netherlands from 1990 to 1992 [38]. This study was performed as part of the Netherlands National Reuse of Waste Research Programme (NOH), funded by the Dutch Ministry of Economic Affairs and the Directorate-General for Environmental Management of the Ministry of Housing, Physical Planning and Environment.

1.6 STRUCTURE OF THIS THESIS

In Chapter 2 a methodological framework for LCA is proposed which distinguishes five different components: goal definition, inventory, classification, valuation and improvement analysis. For historical reasons, this framework differs slightly from that of the SETAC "Code of Practice" [43]. The framework proposed in this thesis was developed before the "Code of Practice" was drafted and in fact served as one of the inputs for the framework discussions of the LCA-experts who drafted the SETAC "Code of Practice".

As far as is possible, the SETAC "Code of Practice" framework and terminology is followed in this thesis. In Chapters 2, 3 and 4, however, this is not feasible because, as mentioned earlier, these chapters were written and published before the SETAC framework was developed and published.

In Chapter 2, the goal definition and inventory components are discussed in detail. Goal definition is concerned with defining the goal of the study in relation to the intended application. Application always involves some kind of comparison, for which a unit of use should be specified that is to form the basis for comparison. The unit is based on the function of the products to be compared, and is called the functional unit. In the inventory component the life-cycle of a product is the guiding concept. In order to make a quantified survey of the environmental inputs and outputs of a product system, the boundaries between the product system and the environment and the boundaries between the product system and other product systems must be determined, and some cut-off point set for the infinite regression of processes needed to produce inputs for other processes. Proposals on how to handle these issues are developed in this chapter.

In Chapter 3 the classification, valuation and improvement analysis components are discussed in detail. In the classification component, the resource extractions and emissions associated with the life-cycle of a product are translated into contributions to a number of environmental problem types, such as resource depletion, global warming, ozone depletion, acidification, and so on. To this end, each extraction and emission is multiplied by a so-called classification factor (or equivalency factor according to SETAC terminology) and the multiplication results are aggregated for each problem type. Equivalency factors are proposed for a number of environmental problem types. Valuation covers both valuation of the different environmental problem types and assessment of the reliability and validity of the results. Methods for these two valuation steps are discussed. Improvement analysis identifies options for improving the product(s) studied. Two complementary techniques for identifying such options are discussed.

In Chapters 4 and 5, new methods are proposed for equivalency factors for human toxicity and ecotoxicity as well as for abiotic and biotic depletion.

In Chapter 4, it is proposed to aggregate potentially toxic substance emissions in one score for human toxicity and two scores for ecotoxicity. This aggregation is based on the multimedia environmental models of Mackay, which simulate the behaviour of substances in the environment, and on toxicity data such as the *acceptable* or *tolerable daily intake* (ADI, TDI) and *no observed effect concentration* (NOEC) of individual substances. It is proposed to render these multimedia models suitable for application in product LCAs by adopting the concept of a reference substance, as used in the ozone depletion potential (ODP) and the global warming potential (GWP).

In Chapter 5, a proposal is developed for equivalency factors indicating the relative

depletion of a resource per unit extracted. It is proposed to distinguish between abiotic and biotic resources and to measure depletion by physical data on reserves, annual production and - for biotic resources - regeneration. Equations are developed for calculating equivalency factors for these two categories of resources, resulting in so-called *abiotic depletion potentials* (ADP) and *biotic depletion potentials* (BDP). ADP- and BDP-values are provided for a number of resources and an illustration given of how they might be applied in LCAs.

In Chapter 6, a case study has been performed to assess the extent to which the results of an LCA are influenced by choices of methodology and data used in the various steps of LCA and also to illustrate the LCA methodology as proposed in the preceding chapters. Four different margarines used for frying, roasting and spreading on bread are assessed and compared. In order to determine the influence of choices of methodology and data, sensitivity analyses have been performed with respect to: allocation methods for coproduction; estimates of pesticide emissions; methods to characterize extractions of abiotic resources; data sets for Global Warming Potentials and Photochemical Ozone Creation Potentials; valuation methods; and impact assessment methods.

In this thesis a number of international developments in the field of LCA methodology are described. The main contributions of the thesis to these developments are the proposal for the methodological framework, the problem-oriented design of the steps of the classification ("Environmental Themes Approach") and the proposals for a number of specific equivalency factors for characterizing extractions and emissions. The methodological framework proposed has served as an important input for the SETAC framework discussions and has been accepted with the adaptations mentioned above. The problem-oriented design of the classification and particularly the proposals for equivalency factors appear to be contributing to the international scientific debate about this topic.

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1.8 APPENDIX 1

LCA workshops and other forums organized since 1990:

Location	Date	Under the auspices of
Washington D.C., USA	May 1990	World Wildlife Fund and the Conservation Foundation
Vermont, USA	August 1990	SETAC
Leuven, Belgium	September 1990	Procter & Gamble
Leiden, Netherlands	December 1991	SETAC
Sandestin, USA	February 1992	SETAC
Washington D.C., USA	March 1992	SETAC
London, UK	June 1992	CESIO
Potsdam, Germany	June 1992	SETAC
Wintergreen, USA	October 1992	SETAC
Sesimbra, Portugal	April 1993	SETAC
Amsterdam, Netherlands	June 1993	UNEP
Leiden, Netherlands	February 1994	SETAC
Brussels, Belgium	April 1994	SETAC

Chapter 2

Quantitative life cycle assessment of products
1. Goal definition and inventory

Introductory note

As mentioned in Chapter 1, there may rise confusion due to the different methodological framework and associated terminology used in the SETAC "Code of Practice" and the framework and terminology used in particularly this chapter. To avoid this confusion as much as possible, an overview of the main differences is provided here. This is followed by a comparison of the framework and terminology used in this chapter and that in the SETAC "Code of Practice".

The framework in this chapter consists of five components. The "Code of Practice" consists of four components. The main difference concerns the components *classification* and *valuation* in this chapter. These are part of the *impact assessment* in the SETAC framework. Classification as used in this chapter is subdivided into *classification* and *characterization* in the SETAC "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 assessment 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"	this thesis
goal definition and scoping	goal definition
inventory analysis	inventory analysis
impact assessment: classification	classification
impact assessment: characterization	classification
valuation	evaluation
improvement assessment	improvement analysis

Guinée, J.B., H.A. Udo de Haes, G. Huppes, 1993. *Quantitative Life Cycle Assessment of Products: 1. Goal definition and Inventory*. **Journal of Cleaner Production** Vol. 1, No. 1, 3-13.

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2.1 ABSTRACT

Quantitative environmental life cycle assessment of products can become a useful tool in product-oriented environmental management. With this methodology the environmental impacts of the product during its entire life cycle are attributed quantitatively to the functioning of the product as far as possible. Currently, the scientific basis of methods for assessing the environmental impacts of products is not yet adequate. Methods are divergent, yield conflicting results and contain considerable gaps. In two successive articles an overview of the similarities and differences between these methods, as developed in different countries, is given. To enable fruitful discussions on methods used, and to make life cycle assessment (LCA) an acceptable tool for product-oriented environmental management, a general methodological framework is proposed. In this first article a general introduction to LCA is given, a general methodological framework is proposed and two components of the methodological framework, the goal definition and the inventory, are discussed in more detail.

2.2 INTRODUCTION

The principle of life cycle assessment (LCA) as a tool for product-oriented environmental management, has become widely accepted, both in Europe [1] and in the US [2]. LCAs can be applied as a tool for decision making. Major categories of possible applications are product improvement, the design of new products, product information, ecolabelling, and the exclusion or admission of products from or to the market. In its most comprehensiive form LCA aims to quantify all environmental impacts of a product during its entire life cycle [3,4,5,6,7]. The life cycle starts with resource extraction, goes through all production steps, the use of the product and ends with the waste treatment. It includes the necessary transportation, recycling and reuse. This is sometimes described as *cradle-to-grave analysis*. The terms LCA and cradle-to-grave analysis indicate that it is not the products per se that are analysed, but in fact *product systems* in the sense of production-consumption-waste treatment systems [8]. However, the function of the product as it is used remains the point of reference to which the environmental impacts are attributed. Products may be any items produced. Attention is mainly given to consumer products like packaging, light bulbs, detergents, etc. But LCA may also be applied to services like car rental, storage, telephone service etc., and policy strategies.

Besides the quantitative LCA covering the entire life cycle of a product, there are also simpler approaches for the environmental assessment of products. Thus qualitative criteria may be used, such as recyclability, repairability, types of resources used, and types of substances emitted, and/or the attention may be focussed at a limited part of the life cycle. Examples of these streamlined approaches can be found in the ecolabel systems of Germany, Japan and Canada. Such methods are relatively fast and simple because of the limited amount and quality of the data needed.

Quantitative LCAs lack this advantage of simplicity. However, quantitative studies may lead to quite surprising results. An example here is milk packaging. Based on the recyclability criterion, it would be expected that returnable packaging used several times is more environ-

mentally sound than all its one use alternatives. But this does not always appear to be true. A quantified comparison of the reusable bottle with the polyethylene milk bag has shown the latter to be better or at least as environmentally sound in all types of impact considered [4,9]. In practice a quantitative approach may thus validate the results of other assessment methods.

Unfortunately, current quantitative LCAs often yield mutually conflicting results. The discrepancies are related to both the research methods and the data used. Thus one quantitative study will favour the reusable milk bottle [5], another the milk carton [3], while still another study will show the scores of the two types of packaging to be similar [6].

In several countries projects are being undertaken to improve the methodology of quantitative LCA, for example, in Germany [10,11], the Netherlands [12] and the US (for the United States Environmental Protection Agency). Data gathering is being undertaken in Switzerland [13], Austria [14] and in an ongoing project commissioned by the Plastic Waste Management Institute (PWMI) in Brussels. These projects are all set up differently using different terminologies. To enable fruitful discussions on methods and data needed, and to be able to apply LCA as an acceptable tool for product-oriented environmental management, there is a need for a general methodological framework.

This article is the first of two articles dealing with the methodological aspects of quantified environmental life cycle assessment. In this article, first a general set-up is discussed as proposed at a workshop for LCA experts, organized under the auspices of the Society of Environmental Toxicology and Chemistry (SETAC) [15]. Then two of the different components of the method, which can be distinguished, are reviewed in more detail: the goal definiton and the inventory. This review includes discussions of similarities and differences between several authors. The article ends with a case study, analysing the causes of the different outcomes of a set of comparable studies. In this case study one of the current classification approach is applied, which will be described briefly in a separate section. In a subsequent article we will discuss the other components of the methodology in detail: classification, valuation and improvement analysis.

2.3 SET-UP OF THE LIFE CYCLE ASSESSMENT

Within a quantitative environmental LCA, five components can be distinguished [12,15]:

1. the *goal definition* of the LCA-study;
2. the *inventory* of all the different types of inputs from and outputs to the environment (environmental inputs and outputs) during the entire life cycle of a product resulting in the *inventory table*;
3. the *classification* converting environmental inputs and outputs into contributions to environmental problems, resulting in the *environmental profile* of a product;
4. the *valuation* of the different elements constituting the environmental profiles, thus substantiating a final appraisal; and
5. the *improvement analysis*.

Roughly the same structure, in different terms, was already proposed by Guinée *et al.* in 1990 at

an LCA workshop in Leuven [16]. The distinction between the methodological components largely depends on the type of expertise which is needed. In the goal definition discussions take place between different participants such as commissioners, consumers and LCA scientists, and technological information is needed about product alternatives that can be significantly compared to each other in relation to the goal of the study. The inventory is pre-eminently a subject of systems analysis theories and process technology. The classification is based on environmental sciences, while the valuation is a subject of social sciences (e.g. decision theory). The improvement analysis is based on applied mathematics [17] and knowledge about process technology.

As suggested by Osnowski and Rubik [18], environmental LCA might be seen as part of a more comprehensive assessment of products including environmental, consumer safety, cost and other aspects. Figure 1 shows the general structure of environmental LCA as part of a comprehensive product assessment including environmental, consumer safety, cost and other aspects. The first component of such a broad approach, the general goal definition, specifies the role of the different assessment lines and is distinguished from the goal definition component as part of the environmental LCA. The same holds true for the general valuation and the valuation component as part of the environmental LCA.

In the general valuation the results of the different assessment lines are weighted against each other. One of the inputs for this weighting is the conclusion of the environmental LCA, which can be based on the results of each of the methodological components mentioned. The general valuation is distinguished from the environmental valuation in which different environmental aspects may be weighted against each other. The application should be regarded as a separate entity, outside the environmental LCA and other assessments as decision support systems. The application is the factual decision(s) made by a company, a public authority, or a consumer. In this article, attention will be paid only to the components of the environmental life cycle assessment of products. These components will be dealt with subsequently. Hereafter, the abbreviation LCA means 'environmental LCA'.

2.4 GOAL DEFINITION

In the first part of an LCA the goal of the assessment is determined: does it refer to the improvement of a given product, the design of a new product, the publishing of product information, the granting of an ecolabel, or the exclusion or admission of products from or to the market? The type of intended application will influence the sequence of and the choices to be made within the different components.

In each of the applications some kind of comparison is made, i.e. a comparison of a product before and after improvement (redesign), a comparison of several design alternatives of a new product, or a comparison between different products with the same function. Then a unit of use should be specified as a basis for comparison. It would be incorrect, for example, to compare one milk bottle with one milk carton because the former is used many more times than the latter. So comparisons may be made in terms of 'the packaging of 1000 l milk', 'the production of 1000 lumen of light', '1000 x the measurement of fever', etc. We cannot simply compare a livestock fodder containing copper with a fodder without copper since the latter yields less meat: 'the production of 1000 kg of meat' would be a valid standard of comparison. Sometimes it is the

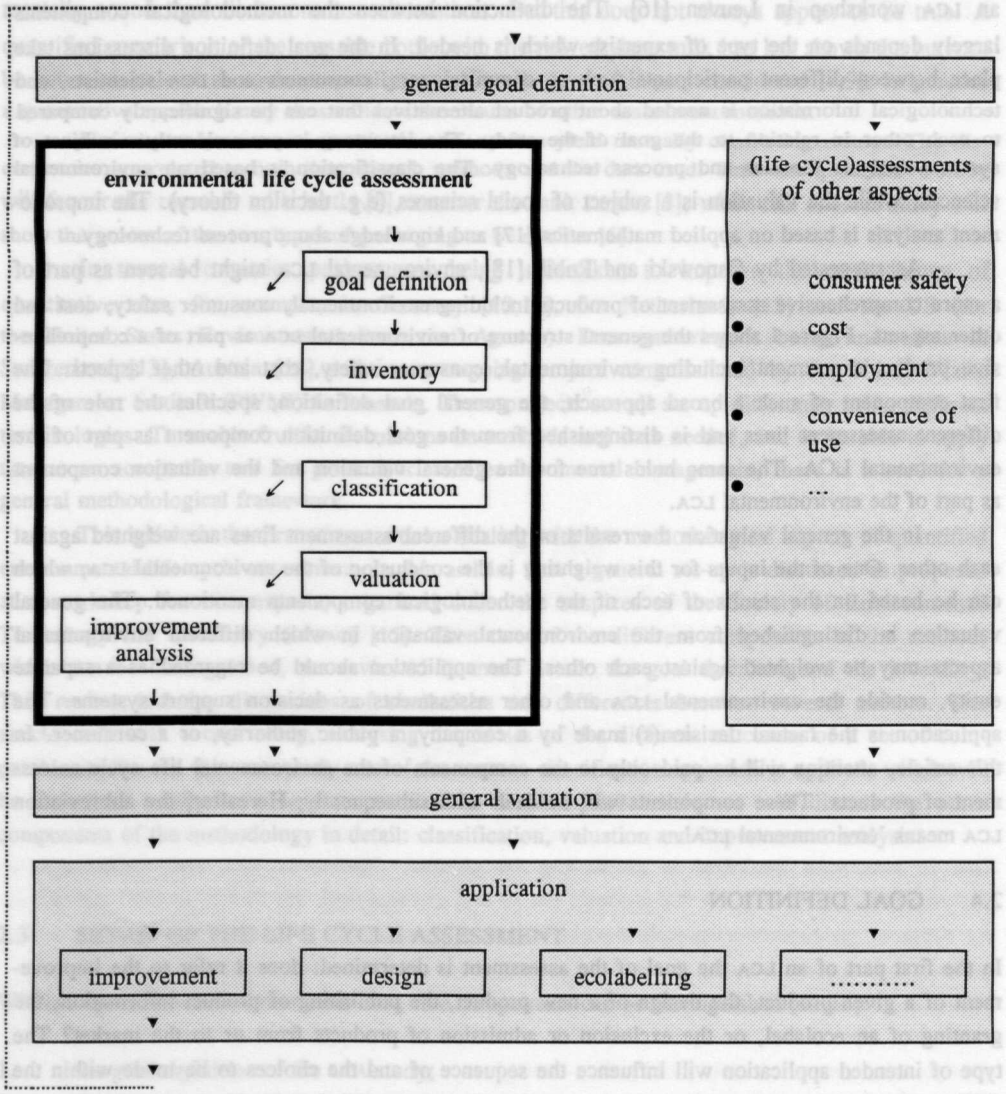

Figure 1: General structure of an environmental life cycle assessment as part of a comprehensive product assessment [22]. (NB not all possible feed back loops are drawn)

duration of use that is the basis for comparison. Lindeyer et al. [19] compared window frames on the basis of '50 frame years'.

We call such units of use which form the basis for comparison and appraisal *'functional units of a product'*. These functional units should then be translated into physical consequences for the products system. For example, the functional unit 'the packaging of 1000 l milk' implies for

the comparison of milk bottles and milk cartons, that 40 milk bottles of approximately 600 g, which are reused 25 times, are compared with 1000 milk cartons consisting of approximately 20 g of carton and 5 g of polyethylene.

Finally, in this first component the spatial scale and the time horizon of the assessment have to be determined. This is of special importance for the choice of representative processes and related data. Consequently, this will be discussed in more detail in the next component in relation to the specification of processes.

2.5 INVENTORY

In this part of the method, the boundaries between the product system and the environment, i.e. the inputs from and the outputs to the environment system, and the boundaries between the product system under study and other product systems are specified. In Figure 2 the relations

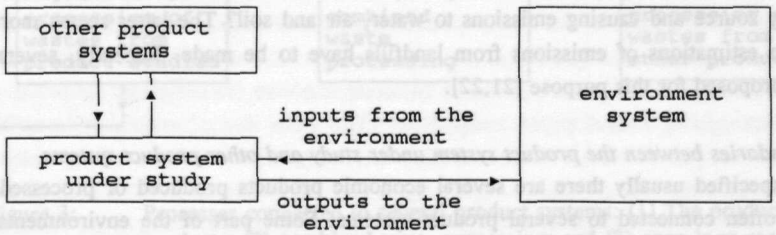

Figure 2: The relations between a given product system, other product systems and the environment

between a product system, other product systems and the environment system is drawn schematically.

In the inventory three elements may be distinguished: the definition of the processes of the product system; the specification of all processes and their data; and the compilation of the inventory tables. In factual studies these elements may be dealt with iteratively, until the results are deemed satisfying.

2.5.1 **Definition of the processes of the product system**

The first element of the inventory is the definition of the product system in terms of all processes necessary for the functioning of a product. In determining the product system, it is important to define what belongs to the product system. Three aspects can be distinguished in this step: definition of the boundaries between product system and environment system; definition of the boundaries between the product system under study and other product systems; and the distinction between relevant and less relevant processes related to the product system under study.

2.5.1.1 *Boundaries between product system and environment system*

There is a double relation between the product system and the environment system. On the one hand there are inputs from the environment through resource extractions and space requirements. In principle, all processes of resource extraction are to be taken into account. Starting with already processed resources such as alumina, or even with the import of purified materials, is at variance with the cradle-to-grave approach and clearly underestimates the environmental inputs and outputs [20]. But still there might be ambivalence about what is meant by *resource extraction*: where does the environment system end and where does the product system begin? Is a growing wood a resource, which may be taken from the environment, or is it a production process which needs inputs in the form of space, machinery, fertilizer, pesticides, etc.? On the other hand there are outputs to the environment. These raise similar questions. In principle the output consists of emissions of substances, noise, ionizing radiation, final waste, etc. The system boundaries may be ambivalent especially in the case of waste. Is waste in a well kept waste dump to be considered as an output to the environment? Or is it to be considered as long-term waste processing (thus as part of the product system), requiring space for the storage of the final solid waste, producing methane as a potential energy source and causing emissions to water, air and soil? The latter seems more appropiate, and then estimations of emissions from landfills have to be made. Recently, several methods have been proposed for this purpose [21,22].

2.5.1.2 *Boundaries between the product system under study and other product systems*

When a process is specified usually there are several economic products produced or processed. These products are often connected to several product systems. Some part of the environmental inputs and outputs then have to be allocated to one product system, another part to a second or third product system. In relation to this allocation three types of processes can be distinguished:

1. the allocation of environmental inputs and outputs of a process producing different economic products, i.e. *production of co-products* (including by-products);
2. the allocation of environmental inputs and outputs of a process processing different waste flows, i.e. *combined waste processing*; and
3. the allocation of environmental inputs and outputs related to *open-loop recycling* (including reuse and recovery).

These three types of multiple-processes are schematically drawn in Figure 3.

Processes usually have a number of economic outputs, (situation 1 in Figure 3), as well as a number with none or even a negative economic value. The latter may be processed as waste in a combined waste processor (situation 2 in Figure 3), in an individual waste processor, or it may be recycled or reused in the same, i.e. closed-loop recycling, or another product system (situation 3 in Figure 3). Closed-loop recycling does not need an allocation because all inputs and outputs belong fully to the product system under study and can be accounted for on that level.

The allocation should be independent of the product which is primarily under study. Basically, it can be done in at least three different ways: in proportion to a common physical unit of the products; in proportion to the economic value of the products; or in proportion to the functions or services delivered by the products. The question is now which approaches may be

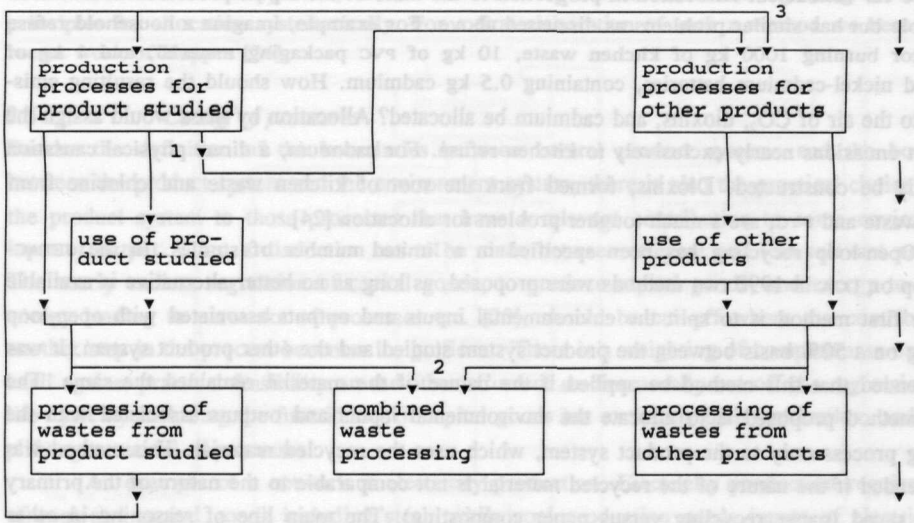

Figure 3: Processes connected to several product systems: (1) The production of co-products; (2) combined waste processing; and (3) open-loop recycling

used in relation to the three types of allocation problems as distinguished above?

The production of co-products has been dealt with in several ways. In the studies conducted so far there are a number approaches based on certain physical quantities such as mass and energy. The units explicitly used to allocate environmental inputs and outputs associated with processes producing co-products are: kg (most studies), moles [10], and energy content [23]. The choice of the unit strongly influences the allocation result. For example, the electrolytic production of chlorine, one of the raw materials used in the manufacture of PVC, co-produces caustic soda and hydrogen. Huppes [24] calculated that, if the environmental inputs and outputs of this process were allocated in proportion to the number of molecules (moles), 25% of the environmental inputs and outputs of the process of electrolysis would have to be allocated to the production of chlorine. If the environmental inputs and outputs were allocated in proportion to the mass of chlorine, the caustic soda and the hydrogen, the figure would rise to 46%. The economic value of the products has been applied as a possible basis for allocation by Basler and Hofman in 1974 [25]. Based on their data we calculated that if the environmental inputs and outputs of the same chlorine process were allocated in proportion to the relative economic value (at that time) of the products, the figure for chlorine would be 36%. This figure would rise to 50% based on long-term average data on proceeds [24]. The third option, allocation in proportion to the function of the products, has been proposed as a possible basis for allocation by Frischknecht *et al.* [26]. In fact, this approach has often been applied, for example, by Boustead [8]. He used the display area in m^2 as a basis for allocation in the retail sale of products, which seems a valid basis for

processes for which space is the limiting factor.

Combined waste handling processes have not yet been dealt with in a specified manner in studies so far conducted. Allocation in proportion to the mass of the inputs processed is technically possible but has similar problems as discussed above. For example, imagine a household refuse incinerator burning 1000 kg of kitchen waste, 10 kg of PVC packaging material, and 1 kg of discarded nickel-cadmium batteries, containing 0.5 kg cadmium. How should the resulting emissions into the air of CO_2, dioxins, and cadmium be allocated? Allocation by mass would assign the cadmium emissions nearly exclusively to kitchen refuse. For cadmium, a direct physical causation can easily be constructed. Dioxins, formed from the soot of kitchen waste and chlorine from kitchen waste and PVC, are a much tougher problem for allocation [24].

Open-loop recycling has been specified in a limited number of studies. In the SETAC-workshop on LCA in 1990 two methods were proposed, as long as no better alternative is available [2]. The first method is to split the environmental inputs and outputs associated with open-loop recycling on a 50% basis between the product system studied and the other product system. It was recommended that this method be applied if the nature of the material remained the same. The second method proposed is to allocate the environmental inputs and outputs associated with the recycling process only to the product system, which uses the recycled material. This method was recommended if the nature of the recycled material is not comparable to the nature of the primary material saved (paper recycling versus paper composting). The main line of reasoning in other methods is to take another process as a reference that produces a product similar to the recycled product. In a study on milk packaging such an approach was developed in practice [5]. Part of the primary material 'saved' was subtracted according to the level or quality of recycling. Without such a differentiation LCA could never provide arguments for cascaded recycling.

To give an example, white glass may be recycled as white glass, as coloured glass, or as filling material in road building. This could be incorporated in the LCA inventory by, for instance, subtracting 25%, 50% and 75% from the hypothetical primary glass production that could replace the recycled amount. The value of the recycled material or component clearly plays a role here. This approach could be generalized for 'open loop' recycling as suggested by Ecobilan [27]. Still, it seems rather unsatisfactory to imply a process in the inventory that does not belong to the product system analysed.

Summarizing the above findings we arrive at the following conclusions on the problem of allocation. Allocation in proportion to some physical unit is rather easy to apply, but the results of such an approach are strongly influenced by the choice of the physical unit. The second possibility, an allocation in proportion to economic values has as an advantage that it reflects the social basis for the functioning of all processes. On the other hand, it requires a great many price data, which themselves are variable. Such data are less stable over time than technical relations. The third possibility, an allocation in proportion to functions, is often used implicitly, disguised as a physical approach. The area in square meters of storage room required for retailing in a warehouse may be the basis for allocating the share of a product in a warehouse. The dimension is *storage room*, which is a functional description, but the allocation unit might be the physical *square meter*.

The example of *storage room* shows that the function and physical approach can be closely related. The same holds true for the function and the value approach. The value,

quantified in financial terms, is always based on the function a good or service has for the consumer acquiring it. Thus, it is not so much a choice between either the physical, the functions or the economic value approach, as a need to develop a reasoned method guiding the choice of principles for allocation systematically. For a first elaboration of such a method we here refer to recent work of Huppes [24].

2.5.1.3 *Cut-off of processes*

Besides the definition of the boundaries between several product systems and the boundaries between the product system and the environment system, there is also the practical choice limiting the product system to those processes that have a relevant contribution to some environmental input or output. Thus, a distinction has to be made between relevant and less relevant processes. The nearly endless regression of capital goods, to produce capital goods, and energy to produce energy leading to networks of processes etc. [20], should be cut off where the contribution of another step in the process becomes insignificant. There is a weighting of completeness on the one hand against practical feasibility on the other. Current studies usually cut off the regression in the life cycle by including the functioning of capital goods but excluding their production, and/or simplifying networks of processes.

An example of a network is electricity production. Electricity production requires inputs of coal and capital goods including steel, while coal production needs inputs of electricity and capital goods including steel. Boustead [20] calculated that in the case of electricity production exclusion of the production of capital goods and simplification of networks to pseudo-linear sequences can introduce significant errors. The inclusion of these two aspects lowered the overall energy efficiency from 0.30 to 0.27, measured as the ratio of the electrical energy delivered to the consumer compared with the energy extracted as primary fuel from the earth. The consequence of this shift is that energy production then contributes relatively more to the total environmental impact of a product system.

In Figure 4 a product system is schematically drawn with respect to the different phases of the life cycle including the boundaries and regression as discussed above.

2.5.2 **Specification of all processes and their data**

When specifying the processes of the product system and their data, one major question is which of several applicable processes to choose. Here both the scale of the comparison and its time horizon are important for guiding the search for the relevant processes. The inventory may refer to the products of one or more specific companies, or it may refer to a product type in a region, such as a country or the EC. If the assessment is meant for a comparison within one or between different companies, company specific technologies should be considered. If, however, the comparison is between products in some region (e.g. the milk bottle in Europe), technologies should be considered which are representative of that given region. Consequently LCAs performed for a company to company comparison cannot be used in a comparison between products in one country, in Europe, or the Western world, and vice versa.

An example may help to clarify this point. The LCA on milk packaging conducted by Lundholm and Sundström [3], at the request of TetraPak Sweden, is based on the Swedish technique of generating electricity and a paper production technique used by this specific company in

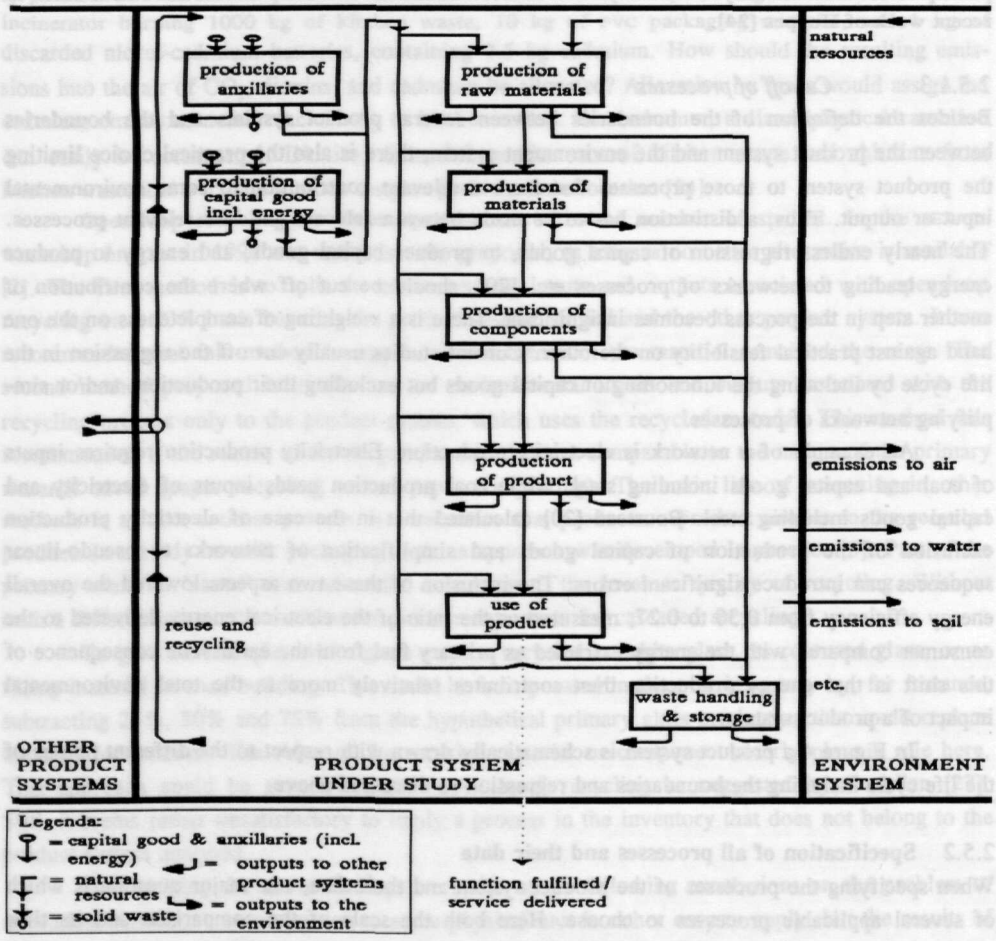

Figure 4: An example of a product system including boundaries between the product system and the environment, and between the product system under study and other product systems

Sweden. This assessment can be used in a product comparison between companies, but should not be used automatically in a comparison of milk packaging systems within individual countries or within the European Community.

Another choice is that of time horizon. It determines the type of the technology to be specified. Is it current products that are to be compared? Or products designed now for marketing

halfway the 1990s? In the LCAs performed so far the choices made do vary. Often current data are sought. But sometimes the data stem from different periods and are rather old. Thus the data in the study conducted by Hunt *et al.* [7] may well have been to date in 1974. Whether this was the case cannot be ascertained. The study conducted by BUS in 1984 [4] used some of the same data as the study by Hunt. Certainly these data certainly were no longer up to date in 1984.

Given a well-defined time horizon a related question is which type of technology is representative for a given period: the worst, the average, the modern, the best practicable means or even the best technical means available? Given that a modestly future-oriented comparison should be made, a reasonable choice would be to take *modern technology* as representative. i.e. the technology which is currently most commonly installed in the area under study. The case study presented at the end of this paper will deal in more detail with the effects of the choice of process technology on the resulting environmental inputs and outputs.

2.5.3 Compilation of the inventory tables

On the basis of the specification of all processes of the life cycle, the environmental inputs and outputs of each process of the life cycle can be quantified. The result of this quantification is the first item that may be used in an environmental assessment of products; the table of inputs from the environment and outputs to the environment per process called *the inventory table per process*.

Finally, the environmental inputs and outputs can be added per type of input and output over the processes related to the whole life cycle of the product under study. For instance all SO_2 emissions are added. This step results in an *inventory table per functional unit of a product* (see Table 2 for an example). The results given in Table 2 are based on a still limited empiricial basis as incorporated in the computer program SimaPro 1.0 [28].

Although the set-up described is of a quantitative nature, also semi-quantitative or qualitative data can be taken into account if these are the only data present. Because it is not possible to add up data on a qualitative or ordinal scale, these data will have to be presented per process and under a separate heading.

2.6 CLASSIFICATION

Here, a brief description is given of the so-called "critical volumes approach", which is a classification method for emissions of substances and has been applied in a number of case studies. This approach is also applied in the case study discussed hereafter. Although quite practical to apply, it has raised a lot of criticism, for a short review see Klöpffer [29]. Thus, improved methods have been proposed [30,31] to be discussed in a future paper.

In the "critical volumes approach" emissions of substances are aggregated per compartment based on environmental standards such as maximum accepted concentrations (MAC values) (these are on-site industry standards). This results in "critical volumes of air" and "critical volumes of water" [4,25] or "units of polluted air" and "units of polluted water" [5,19]. The following is a more precise description of the critical volumes method: emissions are added up after first having been divided by quality standards for human health. Only the standards used do

Table 2: Inventory table of hundred *imaginary* polyethylene bottles (50 gram/bottle) and hundred *imaginary* polyvinylchloride bottles (20 gram/bottle) as computed by SimaPro 1.0.

RAW MATERIALS (kg)	PVC_bott	PE_bottl
brine	$2.0320*10^{+0}$	0
coal	$5.3508*10^{-1}$	0
gas	$1.3544*10^{+0}$	$1.2800*10^{+0}$
oil	$9.5809*10^{-1}$	$5.0786*10^{+0}$
uranium ore	$3.0371*10^{-3}$	0
AIRBORNE EMISSIONS (kg)		
1 _ 2 _ dichloroethane	$3.4000*10^{-3}$	0
CO	$1.1198*10^{-3}$	$1.8900*10^{-3}$
CO2	$3.0202*10^{+0}$	0
Ca	$1.2226*10^{-6}$	0
Cl	$6.0000*10^{-7}$	0
F	$3.3515*10^{-5}$	0
HCl	$2.6000*10^{-4}$	0
Hg	$6.8946*10^{-7}$	0
NOx	$1.3513*10^{-2}$	$3.0500*10^{-3}$
PAH	$2.1576*10^{-9}$	0
SO2	$1.2179*10^{-2}$	$5.4500*10^{-3}$
dust	$9.9683*10^{-4}$	$1.9000*10^{-4}$
hydrocarbons	$2.9172*10^{-3}$	$2.3115*10^{-2}$
vinylchloride	$2.8000*10^{-3}$	0
WATERBORNE EMISSIONS (kg)		
2 _ chloroethanol	$6.0000*10^{-4}$	0
Hg	$3.4000*10^{-8}$	0
Pb	$8.0000*10^{-6}$	0
anorganic suspended substances	0	$1.8800*10^{+0}$
other organic suspended substances	0	$6.0000*10^{-1}$
phenol	$3.8000*10^{-5}$	$2.0000*10^{-5}$
trichloroethanol	$2.4000*10^{-3}$	0
vinylchloride	$1.1400*10^{-5}$	0
SOLID WASTE (kg)		
PVC	$2.0000*10^{+0}$	0
ashes after reuse	$4.6029*10^{-2}$	0
combustion waste	$3.0065*10^{-2}$	0
high active nuclear waste	$1.9910*10^{-8}$	0
low and medium active nuclear waste	$4.8931*10^{-9}$	0
mixed waste (hazardous composition)	$3.0000*10^{-2}$	0
polyethylene	0	$5.0000*10^{+0}$
solid waste	0	$2.0150*10^{-2}$
waste chlorine production	$2.0000*10^{-2}$	0
waste of brine mining	$1.2600*10^{-1}$	0
waste of coal mining	$9.637*10^{-2}$	0

0 = no data given, which may either mean an actual zero or not measured; this is unclear in most data sources.

differ. The Dutch studies [5,19] use MAC values for the aggregation of airborne emissions and EC Directives for surface water intended for drinking water production for the aggregation of waterborne emissions. The Swiss studies [4,25] use German MIC values (maximum immission concentration; "maximale Immisionswerte" des Vereins Deutscher Ingenieure) for the aggregation of airborne emissions, and Swiss Directives for emissions into surface water for the aggregation of waterborne emissions.

2.7 CASE STUDY

Different interpretations and choices related to the methodological elements can have a decisive effect on the final results of a study. To give some idea of this effect, we present a comparative case study of the packaging of milk, which has been the subject of quite a number of LCA studies [3,4,6,32,33,34]. Here we limit our observations to three of these studies, which compared the milk carton and the glass return bottle: a Swedish [3], a Swiss [4] and a German [6] study. The Swiss study together with the work of EMPA [35,36,37,38,39], has been of particular importance in the development of this research area.

The results of these studies were quite divergent: in the Swiss study, the glass bottle scored better than the milk carton on a large number of emissions into the atmosphere and water; conversely, in the Swedish study the milk carton scored better than the bottle on an even larger number of emissions into the atmosphere and water; in the German study, the milk carton scored best for most of the emissions into the atmosphere, while the bottle proved superior for the majority of emissions into water.

One explanation for the discrepancies between the results of these three studies is the fact that the specification of the basis of comparison, the system boundaries, and the classification varied considerably from study to study. Table 3 is a survey of the differences between the three studies for the first two points. With respect to the first point, the basis of comparison of all three studies is the same: 'the packaging of 1000 litres of milk'. Table 2, however, shows that the specification of the product comparison for the glass bottle is different in all three studies. Thus the weight of the bottle in the Swiss research is 400 g, and in the German and Swedish studies, 480 g. The number of trips for the bottle varied in the studies from 10 to 40. The weight of the milk carton in the Swedish and Swiss studies was 20 g of cardboard and 5 grams of polyethylene, while the carton in the German study consisted of 22.5 g of cardboard and 4.5 g of polyethylene.

Table 3: Basic assumptions in two different studies on milk packaging.

	milk carton			milk bottle		
	A	B	C	A	B	C
cardboard/paper (g)	20	20	22.5	1	0	0
polyethene (g)	5	5	4.5	0.4	0	0.12
glass (g)	0	0	0	400	480	370
aluminium (g)	0	0	0	0.7	0.27	0.27
number of trips	1	1	1	20;40	10;20;30	25
demarcation of life cycle	c,e	c,e,t,o	c,t,o	c,e	c,e,t,o	c,t,o

A=Switzerland; B=Sweden; C=West Germany; c=including production, use and waste processes; t=including production processes of transport packaging; o=including distribution and storage processes; e= including production processes for the necessary electricity.

With respect to the second point, the Swiss study was limited to the production of the necessary materials, the cleaning of the bottle and the waste processing of both packages, while the Swedish and German studies also took into account the processing of materials, distribution

and storage processes, and transport packaging. Another discrepancy on this point is that the Swedish and Swiss studies included the emissions of electricity production of all processes involved, while the German study only considered emissions of electricity production related to transport processes.

With respect to the third point, the Swiss study classified emissions for their potential toxic effects on human health and aggregated them in units of polluted air (UPA) and units of polluted water (UPW) as described in the classification section, while the other two studies left out this step.

To make a worthwhile comparison between the results of the three different studies, we drew up new specifications on which to base the comparison of 'the packaging of 1000 litres of milk'. These specifications are based on the Dutch market. The milk carton used there consists of 25.3 g of cardboard and 3.2 g of polyethylene, while the glass return bottle contained 610 g of glass and 0.3 g of aluminium [32]. The average number of trips for the bottle was estimated at 30.

With respect to system boundaries and environmental inputs and outputs we only considered airborne emissions associated with the *production processes* of the materials involved, and the airborne emissions related to the necessary process energy. The procedure involved multiplying the airborne emissions of the various processes by the weight of material needed for the packaging of 1000 litres of milk. It will be clear that for an actual LCA the scope of this particular case study would be far too limited.

Table 4: Results of calculations for the comparison of two forms of milk packaging; airborne emissions are given in UPA per functional unit.

	MAC values mg/m³	1000 cardboard packages			33.3 bottles		
		Swiss	Swed.	BRD	Swiss	Swed.	BRD
- Dust	10	11307	6429	4406	12225	4542	1658
- CO	29	2249	531	1	2935	2646	-
- NH_3	18	-	-	-	389	-	-
- NO_x	9	63144	7491	4	9942	8087	14730
- SO_2	13	182805	32899	15417	16814	23000	7336
- H_2S	15	-	121	540	-	-	-
- HCl	7	-	69	-	-	468	494
- Cl_2	3	-	0	-	-	-	-
- F	2.5	-	-	-	120	892	1487
- C_xH_y	500	560	29	36	76	3	-
- Formaldehyde	1.5	62	-	-	90	-	-
- Ethyleneimine	1	5984	-	-	900	-	-
- Mercaptans	1*	-	531	1771	-	-	-
- Pb	0.15	-	-	-	8	-	-
- Hg	0.01	481	-	-	0	-	-
TOTAL		**266592**	**48100**	**22175**	**43499**	**39638**	**25705**

-=no entry
*MAC value of methylmercaptane

The emissions were subsequently aggregated into UPA according to the provisional model described in the classification section. Table 4 shows the results of these conversions as well as the MAC values by which the emissions were divided [40].

Even now that the basic assumptions for the comparison of the three studies have been equated, there remain substantial discrepancies between the critical air volumes for the carton and the bottle in these studies. Thus, the UPA value of the Swiss carton is nearly six times greater than that of the Swedish carton. The UPA value of the German carton is lower by a factor of two than that of the Swedish one. To explain these differences it is necessary to compare the airborne emissions for the **production** of cardboard, polyethylene, glass and aluminum according to the findings of the different studies (table 5).

Table 5 indicates that, as far as the data can be compared, the Swedish emissions data for the production of cardboard/paper are lower for the entire range than the Swiss data up to a factor of more than 130. The German emissions data are also much lower than the Swedish data or completely absent. This latter difference can be explained by the fact that the German study, as we have seen, does not take into account emissions associated with the generation and use of electricity for the production of materials. Thus, emissions which might be expected from the production of electricity, such as SO_2 and NO_x, are generally extremely low or entirely absent.

Table 5: Emissions associated with the **production** of various materials according to two different sources.

Airborne emissions in mg/kg unless stated otherwise	cardboard/paper			aluminium			glass			polyethene		
	Swiss	Swed.	BRD	Swiss	Swed.	BRD	Swiss	Swed.	BRD	Swiss	Swed.	BRD
- Dust	3900	2430	1630	14400	89100	370	5800	919	810	4500	879	880
- CO	1300	594	-	256600	252000	-	400	55	-	10100	117	10
- NH_3	-	-	-	-	-	-	344	-	-	-	-	-
- NO_x	18200	1730	-	40700	63000	--	3800	2650	6520	33700	7390	10
- SO_2	92300	13300	7910	145700	163000	-	8600	12300	4690	12900	28500	92
- H_2S	-	72	320	-	-	-	-	-	-	-	-	-
- HCl	-	19	-	-	5510	-	-	80	170	-	-	-
- Cl_2 µg/kg	-	10	-	-	-	-	-	-	-	-	-	-
- F	-	-	-	1000	6045	2900	-	20	140	-	-	-
- C_xH_y	6700	50	-	18900	1030	-	1600	66	-	34500	4200	5670
- Formaldehyde	-	-	-	221	-	-	3	-	-	29	-	-
- Ethyleneimine	235	-	-	256	-	-	41	-	-	12	-	-
- Mercaptans	-	21	70	-	-	-	-	-	-	-	-	-
- Pb	-	-	-	4	-	-	-	-	-	-	-	-
- Hg µg/kg	190	-	-	0	-	-	-	-	-	-	-	-

-=no data
0=data available, but $<<1$.

For these reasons it is only possible to compare the Swedish and Swiss studies. Here the *technique* of generating electricity, which varies substantially from one country to another, has proved to be a significant explanation for the differences. Thus 41% of the electricity produced in Sweden came from nuclear sources, 56% is hydroelectric and 3% came from oil [3]. In the Swiss

study, the distribution is based on the US situation in 1972: 48% coal, 17% oil, 24% natural gas and 11% nuclear and hydroelectric. In neither of the two studies were airborne emissions attributed to nuclear energy or hydroelectric generation. As a result, the Swiss airborne emissions for the average production of electricity were much higher than the Swedish emissions. The figures in Table 5 are consequently affected by these differences, to an extent which varies according to the type of emission in question. Thus according to the Swiss study the fraction of the emissions of dust and NO_x which results from the production of electricity in the manufacture of cardboard amounts to about 60% and 21% respectively.

Not only was the technique of generating electricity different but also techniques used in the production of certain packaging materials proved to be different. Thus in the Swedish study cardboard is produced according to the sulphate process, while the Swiss study assumes the sulphite process. The SO_2-emission of the latter is higher by a factor of nearly four. Also the types of substances emitted differ.

Next, there are considerable differences in the age of the data. The Swiss data for glass production are relatively recent (about 1983), whereas the data for cardboard production and the generation of electricity come from the late 1960s. Moreover, the sulphite process used in this study is no longer in current use. The Swiss emissions data for cardboard and polyethylene manufacture and the generation of electricity date from about 1984, while the remaining data go back to about 1975.

In the Swiss study, data on the sulphate process are also provided. To illustrate the potential role of the age of data, Swedish data on the sulphate process can be compared with Swiss data on this process. It then appears that the Swiss emissions data are higher than the Swedish by a factor of up to 180.

The reliability of the data may be a third factor accounting for the substantial discrepancies between the Swiss and Swedish figures. As noted in a previous section, these studies do not make clear whether the emission of a substance has been omitted because it has not been analysed or because it is under the detection limit.

2.8 SUMMARY AND CONCLUSIONS

Two components of a methodological framework have been discussed: goal definition and inventory. In the **goal definition** the goal of the study is defined in relation to the application intended. The type of application will influence the whole procedure. The applications always involve some kind of comparison for which a unit of use should be specified forming the basis for comparison. The unit is based on the function of the products to be compared, and is called *the functional unit of a product*. Also the spatial scale and the time horizon of the study are determined in the goal definition.

In the **inventory** the life cycle is the guiding idea. In order to make a quantified survey of the environmental inputs and outputs of a product system, the boundaries between the product system and the environment and the boundaries between the product system and other product systems have to be determined. The latter is as yet undefined in three cases: production of co-products, combined waste processing and open-loop recycling. Allocation in proportion to some

physical unit is rather easy to apply, but the results are strongly influenced by the choice of the unit. An allocation in proportion to economic values reflects the social basis for the functioning of all processes, but needs a great many price data, which themselves are variable. An allocation in proportion to functions is often used implicitly, disguised as a physical approach. Furthermore, it also has to be determined how a distinction can be made between relevant and less relevant processes related to the product system.

As for the choice of representative processes and process data we suggest that for the comparison of current products, technologies are considered which have been installed recently. Furthermore, the choice of representative processes is determined by the scale of the comparison, for example between firms or within countries.

The inventory tables which result, can be compiled per process or per functional unit respectively, resulting in an inventory table per process (or group of processes) or an inventory table per functional unit of a product. Inventory tables consist only of inputs from and outputs to the environment.

A survey of a number of case studies on milk packaging shows that no study uses explictly specified methods. Studies differ with respect to different definitions of the functional unit, different system definitions, a regionally different choice of processes, different periods of data gathering, different choices of environmental problems to be considered and different factors relating environmental inputs and outputs to environmental problems. Consequently, the results of these studies differ markedly.

The inventory is not yet clear cut. For example, the principles for allocation of environmental inputs and outputs in multiple processes have yet to be agreed. Probably, there will never be a single solution to these allocation problems. At the same time, they may strongly influence the results of a particular study. Therefore, it might be a good idea to strive for a standard sensitivity analysis, to be applied in future case studies and to be based on all significant allocation methods developed.

Another problem that needs to be solved is the availability and the quality of process data. There is the current danger that, while interesting methodological progress is being made, the basic data needed are not always available or of good quality. Quality assessment methods should be developed to improve this aspect. Approved data should preferably be stored in publicly accessible databases, which are regularly updated.

Clearly, these problems need to be adressed if life cycle assessment is to make a genuine contribution to the solution of environmental problems as expected.

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Chapter 3

Quantitative life cycle assessment of products
2. Classification, valuation and improvement analysis

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3.1 ABSTRACT

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

3.2 INTRODUCTION

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

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

discussed in detail in the first article. A short summary of the content of these two components is given below.

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

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

The general question addressed in this article is how to proceed after the inventory. This topic has received a lot of attention lately [2,3,4,5,6,7,8,9], but it has also caused extensive discussions. Thus, maybe another question should be asked first: why should in LCA any further components be included at all? The major reason is that emissions (of substances) and extractions (of resources) as listed in the inventory, have no meaning in themselves. It is the problems caused by these extractions and emissions which are important. Basically it is true that, if in a comparison the data on all emissions and extractions point in the same direction, further analysis of these problems is not necessary to reach a decision. In most cases, however, one product is better on some extractions and/or emissions and worse on others; and the same may hold true for product improvement. Then further information about the relation between the extractions and emissions and the environmental problems is needed.

In this additional information both knowledge about environmental processes and effects and social weighting processes play a role. Now two approaches are possible. The first is that these two elements are combined in one single methodological component as is proposed by Ahbe *et al.* [2], by Ryding [4] and by Krozer [10]. In these methods all environmental inputs and outputs of the inventory table are aggregated in one step into one overall score. The second approach is that the two elements are separated and dealt with in two successive components, i.e.

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yet.

the classification and the valuation. In the USA, classification and valuation are described under the heading impact analysis [11]. One of the main arguments for this separation is that each element needs its own expertise. Thus, in the *classification* extractions and emissions are aggregated per type of environmental problem, applying as much as possible scientific knowledge about environmental processes and effects. Or in other words, in the classification extractions and emissions are aggregated on the basis of their potential effect on a number of assessment endpoints (problem types). In the *valuation* different problem types are weighted against each other based on social values and preferences. As already presented in the previous article, we propose to distinguish these two components.

3.3 CLASSIFICATION

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

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

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

3.3.1 Methods for classification

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

methods for classification of emissions of substances were proposed:

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

This seems to give a large choice in classification methods. However, looking in more detail at these five methods, the choice proves to be smaller. In particular, the *loading assessment* does not meet any of the elements of the classification definition. In this method the principal 'less is better' is applied without an assessment of the different environmental effects of the inputs and outputs. In fact, it concerns a grouping of the data of the inventory table without further analysis. A *site-specific exposure-effect assessment* on the other hand is not practicable in an LCA which generally is about dozens of processes all over the world. This method is more appropriate in an *environmental impact assessment* (EIA), where an environmental analysis generally is performed for one activity at a well-defined site. Consequently, only the *impact equivalency assessment*, the *toxicity, persistence and bioaccumulation profile approach* and the *generic exposure-effect assessment* are left. An *impact equivalency assessment* only deals with potential effects on endpoints without regarding preceding environmental processes. Examples of this approach are the critical volumes approach as mentioned above, and the approaches for acidifying and nutrifying emissions (see below). Example of a *toxicity, persistence and bioaccumulation profile approach* is the Swedish proposal for the assessment of ecotoxic substances [8] which includes an assessment of the inherent toxicity by means of an LC_{50}, an assessment of the bioaccumulation by means of a so-called *bioconcentration factor* and a qualitative assessment of the persistency of a substance by classifying them into 'readily' or 'not readily biodegradable'. This results into four partial effectscores for ecotoxic substances [8]. Examples of the *generic exposure-effect assessment* are the classification of ozone depleting emissions according to *ozone depletion potentials* (ODP), the classification of greenhouse emissions according to so-called *global warming potentials* (GWP), and the classification of photochemical oxidants creating emissions according to so-called *photochemical ozone creation potentials* (POCP). These methods result into one general effectscore per problem type.

The *impact equivalency assessment*, the *toxicity, persistence and bioaccumulation profile approach* and *the generic exposure-effect assessment* are three possible methods for classification of emissions of substances. We think that the *generic exposure-effect assessment* resulting into one

effectscore per problem type is the preferred method for LCA, while the *impact equivalency assessment* and the *toxicity, persistence and bioaccumulation profile approach* can offer a temporary solution as long as a *generic exposure-effect assessment* is not yet feasible. Problem of the *toxicity, persistence and bioaccumulation profile approach* is of course how to weight the different aspects againts each other.

In the classification four steps can be distinguished:

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

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

3.3.2 Environmental problem types

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

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

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

The possibilities to predict effects decreases as the order of effects increases. In principle, inputs and outputs should be linked to the lowest order effect, which can still be clearly related to the effect chain considered. Thus, global warming in terms of a change in radiative forcing should

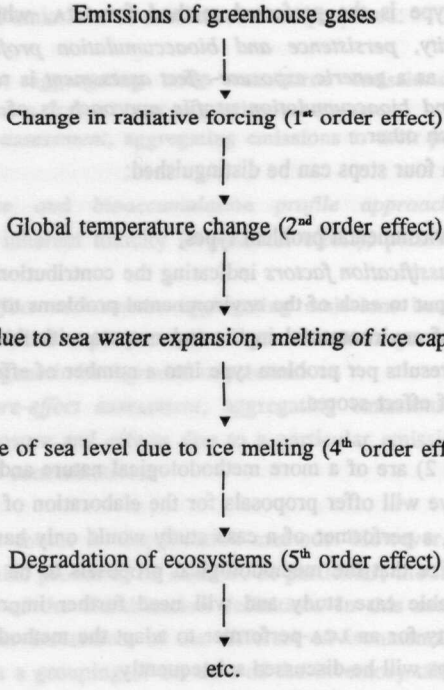

Figure 1: Environmental effect chain for global warming [cf. 8].

preferably be chosen as the assessment endpoint of the classification, although it is only an intermediate point in the effect chain for global warming.

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

Table 1: Generally recognized environmental problems

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

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

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

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

categories of problem types.

3.3.3 Definition of classification factors

3.3.3.1 *Depletion*

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

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

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

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

3.3.3.2 *Pollution*

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

substance.

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

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

For example, for the greenhouse effect we could draw up five subscores: one for substances for which the global warming effects can be quantified by means of so-called *global warming potentials* (in kg CO_2 equivalents), one for the total CH_4 emissions (in kg CH_4), one for the total CO emissions (in kg CO), one for the total NO_x emissions (in kg NO_x) and one for the total emissions of non-methane hydrocarbons (in kg NMHC). In the same way there would be five subscores for ozone depletion and two subscores for photochemical oxidant formation. Hence it would appear to be better and more practical to deal with these uncertainties as such. For example, 'flags' (qualitative remarks) could be attached to substance emissions which may have indirect effects. The values of the associated indirect GWPs and ODPs at which the outcome of the LCA would change could then be calculated in a reliability analysis (see Valuation section). It could

then be considered whether the values calculated are realistic, given the current level of understanding. Substances known to contribute to certain problems but for which classification factors cannot yet be determined could be dealt with in a similar manner.

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

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

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

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

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

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

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

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

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

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

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

In defining the effect part for ecotoxicity, the same problems as mentioned for human toxicity have to be solved. In addition, an assessment of ecotoxic effects has to take into account the large number of species within an ecosystem. A solution for these problems may be found

along the same line as for human toxicity, except that threshold values for ecosystems have to be derived from a number of single species toxicity data. For this several methods have been developed which may be applied to derive these threshold values [33,34,35]. We propose to distinguish between terrestrial and aquatic ecosystems, because of the different species present in these media and the different exposure routes of these species. As specific toxicity data for ecosystems in the sediment and for exposure of ecosystems to air are lacking, these compartments are not yet considered.

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

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

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

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

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

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

its odour threshold value [37].

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

3.3.3.3 *Disturbances*

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

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

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

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

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

3.3.4 Multiplication and aggregation

Third step of the classification is the multiplication of environmental inputs and outputs with their specific classification factors and the aggregation of the results per problem type. It is suggested to

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

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

3.3.5 Normalization of effect scores

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

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

3.4 VALUATION

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

3.4.1 Valuation of the effect scores of the environmental profiles

The classification will result in an environmental profile, which as much as possible is still the product of empirical knowledge about economic and environmental processes. Thereafter, a valuation can be desirable for both product comparison and product improvement. In product comparison the effect scores of the environmental profiles of different products often have to be weighted in relation to each other, while for product improvement a weighting of the effect scores of the product under study is necessary to determine on which aspects the product should be improved primarily. In principle, social values and preferences dominate in this valuation. These

values and preferences could be approximated with policy aims, costs, or with the help of experts or an expert panel and then be the input of a qualitative or quantitative multicriterion analyses. Policy aims and/or costs have proven to be practical indicators in methods which treat the classification and the valuation as a unitary methodological component [2,4,10]. It could be further investigated whether policy aims and/or costs are also appropriate and practical indicators in a separate valuation.

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

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

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

3.4.2 Evaluation of the reliability and the validity of the results

A valuation of environmental profiles without an assessment of the reliability and the validity of

the results, is of little value. The step concerns a sensitivity analysis regarding the influence of both the uncertainty of data and the assumptions and choices made.

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

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

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

3.5 IMPROVEMENT ANALYSIS

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

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

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

For the improvement of products, knowledge of the dynamic behaviour of the environmental profile in terms of process modifications can be even more important. The *marginal*

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

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

3.6 SUMMARY AND CONCLUSIONS

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

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

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

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

here is the compilation of such a standard list, with a sufficiently broad societal basis.

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

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

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Chapter 4

A proposal for the classification of toxic substances within the framework of life cycle assessment of products

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4.1 ABSTRACT

Quantitative life cycle assessment (LCA) is a method allocating the environmental impacts of the whole life cycle of a product to the functioning of that product. The scientific basis of the method is still being elaborated. In this paper a proposal is made to improve the scientific basis of one specific step of the method: the aggregation of potentially toxic emissions of substances in one score for human toxicity and two scores for ecotoxicity. The aggregation is based on multimedia environmental models of Mackay simulating the behaviour of substances in the environment, and on toxicity data such as *acceptable* resp. *tolerable daily intake* (ADI resp. TDI) and *no observed effect concentration* (NOEC) per substance. It is proposed to apply models describing the environmental fate of toxic substances in LCAs of products. In addition, it is proposed to adopt the concept of a reference substance, as used in the ozone depletion potential (ODP) and the global warming potential (GWP), to assess and aggregate emissions of potentially toxic substances.

4.2 INTRODUCTION

The principle of life cycle assessment, abbreviated LCA, as a tool for product oriented environmental management, has become widely accepted, both in Europe [1] and in the USA [2]. An LCA is an analyzing tool for the assessment of the environmental impact of the functioning of a product. That the functioning of a product does not only include the usage of the product, but also the production, transportation, maintenance and waste handling, is reflected in the term life cycle. Among possible applications are the comparison of product alternatives and the (re)design of products in an environmentally optimal way.

Life cycle assessment is a rapidly developing area of applied environmental science. One of the recent activities in the field of LCA is the development of a methodological framework. Within this framework five components may be distinguished: goal definition, inventory, classification, valuation and improvement analysis [3,4,5].

In the *goal definition*, the subject of study is determined. This includes a description of the amount of function investigated, the so-called *functional unit*. An example of a functional unit is "packaging of one sandwich". Using this functional unit, packaging systems of different materials (polyethene, aluminium, paper, etc.) can be compared.

The next component of an LCA is the *inventory*. Within the inventory, the life cycle of each of the products considered is defined by assembling data of the processes which constitute the life cycle. Examples of processes considered are production of materials and components, transport, use and maintenance of products, and waste handling and recycling. Process data consist of economical data (use and production of materials, products and services) and of environmental data (extractions of resources and emissions of substances). The result of the inventory is a list of inputs from and outputs to the environment in terms of extractions and emissions caused by a functional unit of the product studied.

In the *classification*, scientific knowledge of environmental processes is used to estimate the contribution of all extractions and emissions to a limited number of generally recognized environmental problems. For emissions of substances, this is achieved by multiplying emissions

by a classification factor, which is defined per type of problem and per mass unit of a substance emitted. The aim of the classification factor is to provide a scientific basis for the comparison of products on *potential* environmental effects. It is not an indicator of the *actual* effects. Classification and valuation are sometimes treated as a unitary component under the heading impact analysis [6].

In practice one product alternative will seldom be preferred to another one in all environmental aspects. Thence the need for a *valuation*, in which the relative importance of each of the environmental problems is assessed. The valuation facilitates a decision on the choice between product alternatives, or on the subject of product improvement.

One of the applications of LCA is the improvement of products. As the *improvement analysis* demands it own methods and its own areas of knowledge, such as process engineers and technologists with knowledge about processes and materials, this analysis is treated as a separate component in the methodological framework.

For a more comprehensive discussion of the principles and elaboration of these components, we refer to previous publications on LCA-methodology [4,5,7].

4.3 CLASSIFICATION OF TOXIC SUBSTANCES

One of the components of LCA, which has received a lot of attention during the past year, is the classification [8,9,10]. Classification factors have been defined for quite a number of environmental problems: *global warming potentials* (GWPS) [11], *ozone depletion potentials* (ODPS) [12], *photochemical ozone creation potentials* (POCPS) [13], *acidification potentials* (APS) [14], and
nutrification potentials (NPS) [14]. Most of these factors are the results of models simulating the relevant environmental processes in the same way as the GWPS. All factors indicate the contribution of particular emissions to *one* overall mechanism, e.g. ozone depletion.

Based on the general description of the classification given above, the classification of toxic substances should be subdivided into the generally distinguished headings *human toxicity* and *ecotoxicity*. The classification factors for human toxicity and ecotoxicity should as far as possible be science-based, which implies that they should not be based on political standards but on experimental toxicity data and that they should take into account environmental processes such as biodegradation and dispersion, which can influence exposure significantly.

Some studies [15,16,17] aggregate toxic substances according to the so-called "critical volumes approach". In this approach, emissions are aggregated based on occupational health standards and drinking water standards. Since both sets of standards are formed by a compromise between toxicological considerations and technical and economic feasibility, and environmental processes are not included, this approach does not meet the above given description of the classification and needs to be improved.

To improve the assessment of toxic substances following this description, a number of problems have to be faced. For human toxicity and ecotoxicity there is no overall mechanism, such as for ozone depletion and global warming, that toxic substances contribute to. Moreover, ecotoxicity involves not one but thousands of different species that all react differently when

exposed to a particular substance. Several methods to classify toxic substances in different classes have been developed using criteria such as biodegradability/persistency, dispersion in the environment, accumulation, toxic effect in mammals, etc. [18,19]. Other methods aim to provide a quantitative integrated assessment of toxic substances. For this, models are developed relating emissions to exposure concentrations [20,21,22,23], and assessing the potential effects of these exposures [24,25]. Within the framework of LCA it is practically impossible to aim for a site-specific assessment of emissions of toxic substances, taking into account site-specific conditions like the number of people living in the neighbourhood of a factory, the distance between factory and residential districts, the presence of specific ecosystems, the soil composition of that particular site, etc. For such a site-specific assessments other instruments have been developed, e.g. *environmental impact assessment* (EIA). In LCA a product is the starting point of the assessment, which includes an inventory of the emissions of a large number of processes all over the world. It seems practically impossible to gather site specific data for all these processes. In an EIA, the assessment is limited to one or two activities at a specific site allowing a site-specific elaboration of the assessment. In LCA it is thus necessary to abstract from aspects which differ per site and to include these aspects, if possible at all, in a generic way (e.g. in percentages per area).

In this paper a method is proposed to improve the "critical volumes approach" applying the quantitative integrated model approach and deriving formulae for classification factors. The paper is an elaboration of recent work [26]. We propose to distinguish between human toxicity and ecotoxicity and define, similar to the ODP- and GWP-concepts, a so-called HTP (*human toxicity potential*), a TETP (*terrestrial ecotoxicity potential*) and an AETP (*aquatic ecotoxicity potential*) for each substance. With these classification factors an emission of a substance to a compartment $m_{subs,comp}$ can be expressed in terms of an emission of a reference substance m_i:

$$m_i = L_{subs,comp} \times m_{subs,comp} \qquad (1)$$

where $L_{subs,comp}$ is the classification factor of substance *subs* initially emitted to compartment *comp*.

The general calculation procedure for each of these potentials is the same. Each potential in principle exists of two parts: an *exposure* part translating a particular emission to a dose to which a receptor is exposed (human or ecosystem), and an *effect* part translating a particular exposure dose to possible effects on potential receptors. The exposure and the effect part have to be defined for each exposure route and for each substance and the reference substance for the HTP, as well as for the TETP and the AETP. Combination of the exposure and the effect part relative to the exposure and the effect part for the reference substance yields the classification factor, which represents the potential contribution of a unit amount of a given substance to human toxicity, terrestrial ecotoxicity resp. aquatic ecotoxicity relative to a unit amount of a reference substance emitted to a reference compartment.

The general principle of the classification factors, the exposure parts, the effect parts, the combination of these parts into the different classification factors and the necessary further developments will be discussed subsequently.

4.4 GENERAL PRINCIPLE

The exposure part of the classification factor should preferably be based on multi-media environmental models such as developed by Mackay [23]. Until today, these models are mainly applied for the assessment of substances, for example in the Netherlands [24,25]. In the widely used level III models of Mackay, a diffuse emission flux into a predefined standard environment ("unit world") is assumed, leading to a steady-state partitioning between environmental compartments based on processes such as (ad)sorption, deposition, evaporation and leaching, and taking into account degradation processes. In this way equilibrium concentrations are calculated due to emission fluxes. Notice that this implies that high exposure concentrations, possibly occurring before the equilibrium concentration is reached and resulting in acute toxic effects, cannot be considered with these models. Toxicity assessments based on Mackay level III models are thus limited to chronic toxic effects. Mackay models are linear models in the sense that partitioning coefficients and lifetimes of a particular substance are independent from the concentration of that substance.

Mackay models cannot be applied directly to emissions as quantified in LCAs, because LCA-emissions are not restricted to a certain period of time. Emissions of a substance during the product's entire life cycle take place at a non-homogeneous and unknown rate. An LCA is only concerned with the total emission of a substance associated with the entire life cycle of a product, which is regarded as a *pulse* (in kg). Multi-media environmental models, which take into account time-dependent processes such as degradation and partitioning, are necessarily based on a *flux* (in $kg \times day^{-1}$). There is a relation between the flux and the equilibrium concentration. Increasing the flux leads to an increased concentration, and thus to an increased risk.

In principle, two types of solutions for the flux-pulse problem are possible:

- to assume an arbitrary time-period during which the emission takes place;
- to select a reference substance and calculate a dimensionless classification factor per substances similar to the ODP-, GWP- and POCP-concepts.

The first type of solution is not very elegant and can lead to arbitrary results dependent on the time-period chosen. The second solution seems elegant and quite simple to elaborate at the same time and is worked out here. Below, the concept of the reference substance is developed in detail.

When the relation between the flux Φ_{subs} of a substance *subs* and the toxic effect T_{subs} is assumed to be linear, we have

$$T_{subs} = \frac{K_{subs}}{NEL_{subs}} \times \Phi_{subs} \qquad (2)$$

where K_{subs} is an exposure modelling constant, which depends on properties of substance *subs* such as lifetime and partitioning coefficients and the exposure routes, and NEL_{subs} is a no-effect level for substance *subs*, which is regarded as a measure for its toxicity to a specified receptor.

The flux is defined as the mass m_{subs} emitted during some unit time t:

$$\Phi_{subs} = \frac{m_{subs}}{t} \qquad (3)$$

In an LCA the mass emitted is known, but the time period during which the emission takes place is unspecified. To be able to use the modelling equation (2), we will avoid the unknown time t by adopting the concept of a reference substance. For a reference substance *refsubs* one has, similar to (2) and (3)

$$T_{refsubs} = \frac{K_{refsubs}}{NEL_{refsubs}} \times \Phi_{refsubs} = \frac{K_{refsubs}}{NEL_{refsubs}} \times \frac{m_{refsubs}}{t} \tag{4}$$

One can combine (2), (3) and (4) in order to eliminate the unknown t:

$$m_{refsubs} = \Phi_{refsubs} \times \frac{1}{\Phi_{subs}} \times m_{subs} = \frac{T_{refsubs} \times NEL_{refsubs}}{K_{refsubs}} \times \frac{K_{subs}}{T_{subs} \times NEL_{subs}} \times m_{subs} \tag{5}$$

From this, it is possible to calculate the mass of the reference substance, required to cause a toxic effect equal to the toxic effect of substance *subs* caused by the specified emission of m_{subs}. This mass of the reference substance will be denoted by m_l:

$$m_l \equiv m_{refsubs} \text{ such that } T_{refsubs} = T_{subs} \tag{6}$$

and is given by

$$m_l = \frac{K_{subs} / NEL_{subs}}{K_{refsubs} / NEL_{refsubs}} \times m_{subs} \tag{7}$$

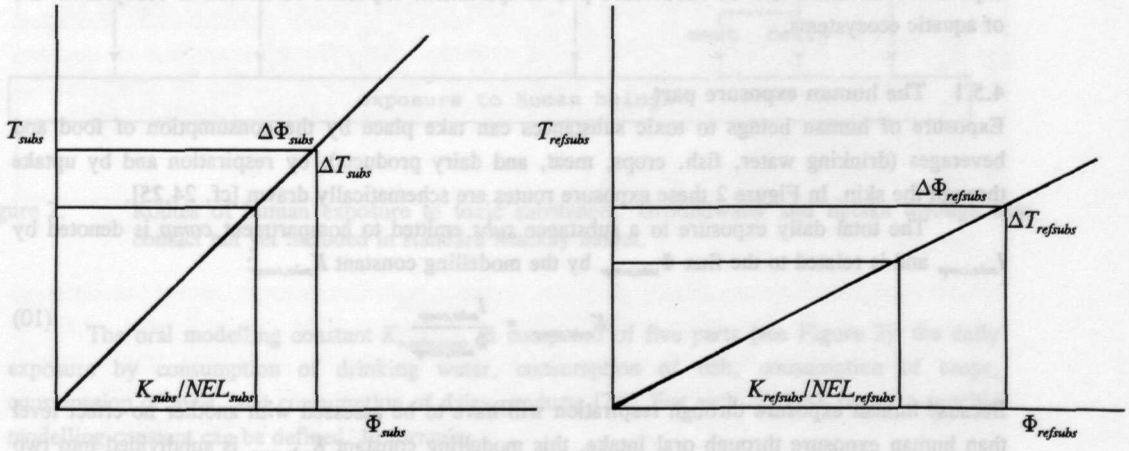

Figure 1: $\Delta\Phi_{refsubs}$ is defined such that $\Delta T_{refsubs} = \Delta T_{subs}$.

This is illustrated in Figure 1. To account for the fact that the eventual exposure concentrations depend on the *initial* compartment the substance is emitted to, equation (7) can be extended with a subscript *comp* denoting this initial emission compartment. The reference substance is thus also

specified in terms of a reference emission compartment *refcomp*:

$$m_l = \frac{K_{subs,comp} / NEL_{subs}}{K_{refsubs,refcomp} / NEL_{refsubs}} \times m_{subs,comp} \quad (8)$$

A final complication arises by the fact that the no-effect levels may differ per intake route. For human toxicity a distinction will thus be made between a respiratory NEL and an oral NEL. This set-up also enables the future inclusion of more routes, e.g. dermal intake. The total toxic effect is found as the sum of the partial toxic effects over all routes:

$$m_l = \frac{\sum_{route} K_{route,subs,comp} / NEL_{route,subs}}{\sum_{route} K_{route,refsubs,refcomp} / NEL_{route,refsubs}} \times m_{subs,comp} \quad (9)$$

Now, m_l denotes the mass of the reference substance emitted to the reference compartment, required to cause a toxic effect equal to the toxic effect of substance *subs* caused by the specified emission $m_{subs,comp}$ to compartment *comp*. It will be clear that the classification factor is dimensionless, and that the classification factor of the reference substance is equal to one.

4.5 THE EXPOSURE PART

Below, the exposure part of the classification factor will be elaborated for human and ecosystems exposure. The latter will be subdivided per compartment: exposure of terrestrial ecosystems and of aquatic ecosystems.

4.5.1 The human exposure part

Exposure of human beings to toxic substances can take place by the consumption of food and beverages (drinking water, fish, crops, meat, and dairy products), by respiration and by uptake through the skin. In Figure 2 these exposure routes are schematically drawn [cf. 24,25].

The total daily exposure to a substance *subs* emitted to compartment *comp* is denoted by $I_{subs,comp}$ and is related to the flux $\Phi_{subs,comp}$ by the modelling constant $K_{subs,comp}$:

$$K_{subs,comp} = \frac{I_{subs,comp}}{\Phi_{subs,comp}} \quad (10)$$

Because human exposure through respiration will have to be assessed with another no-effect level than human exposure through oral intake, this modelling constant $K_{subs,comp}$ is subdivided into two parts: the respiratory modelling constant $K_{r,subs,comp}$ and the oral modelling constant $K_{o,subs,comp}$. $K_{r,subs,comp}$ is defined as:

$$K_{r,subs,comp} = \frac{I_{r,subs,comp}}{\Phi_{subs,comp}} \quad (11)$$

where $I_{r,subs,comp}$ is the daily human exposure by respiration, defined as

$$I_{r,subs,comp} = C_{r,subs,comp} \times \dot{V}_r \qquad C_{r,subs,comp} = C_{air,subs,comp} \text{ (see discussion)} \qquad (12)$$

where $C_{r,subs,comp}$ is the concentration of substance *subs* initially emitted to compartment *comp* in the air respired (route), $C_{air,subs,comp}$ is the concentration of substance *subs* initially emitted to compartment *comp* in the compartment air of the "unit world" and $\dot{V}_r$ is the average volume of air respirated by a human being per day.

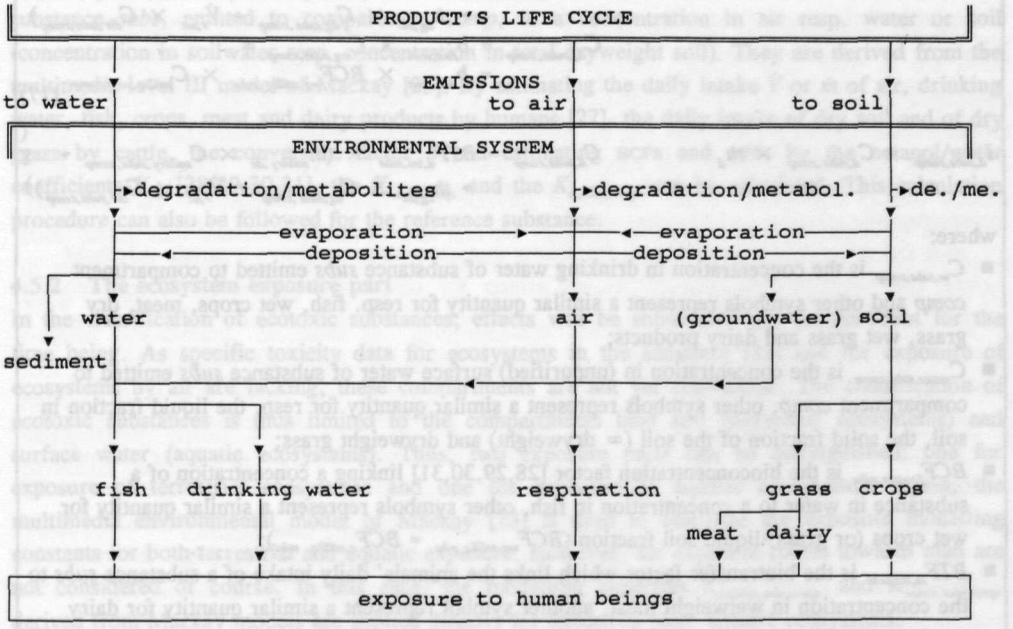

Figure 2: Routes of human exposure to toxic substances. Groundwater and uptake through skin contact not yet included in standard Mackay model.

The oral modelling constant $K_{o,subs,comp}$ is composed of five parts (see Figure 2): the daily exposure by consumption of drinking water, consumption of fish, consumption of crops, consumption of meat, and consumption of dairy products [24]. For each of these routes a specific modelling constant can be defined. In formula:

$$K_{o,subs,comp} = \frac{I_{w,subs,comp} + I_{f,subs,comp} + I_{c,subs,comp} + I_{m,subs,comp} + I_{d,subs,comp}}{\Phi_{subs,comp}} \qquad (13)$$

These partial intakes will be worked out subsequently, largely based on Toet *et al.* [24] and De Nijs & Vermeire [27] (see formula box).

$$I_{w,subs,comp} = C_{w,subs,comp} \times \dot{V}_w \qquad C_{w,subs,comp} = C_{water,subs,comp} \text{ (see discussion)} \tag{14}$$

$$I_{f,subs,comp} = C_{f,subs,comp} \times \dot{m}_f \qquad C_{f,subs,comp} = BCF_{f,water,subs} \times C_{water,subs,comp} \tag{15}$$

$$I_{c,subs,comp} = C_{c,subs,comp} \times \dot{m}_c \qquad C_{c,subs,comp} = BCF_{c,soilliq,subs} \times C_{soilliq,subs,comp} \tag{16}$$

$$I_{m,subs,comp} = C_{m,subs,comp} \times \dot{m}_m \qquad C_{m,subs,comp} = BTF_{m,int,subs} \times (\dot{m}_{soildry,mc} \times C_{soildry,subs,comp} + \dot{m}_{dg,mc} \times C_{dg,subs,comp} + \dot{V}_{r,mc} \times C_{air,subs,comp})$$
$$C_{dg,subs,comp} = b_{dwt,wwt} \times C_{wg,subs,comp} = b_{dwt,wwt} \times BCF_{wg,soilliq,subs} \times C_{soilliq,subs,comp} \tag{17}$$

$$I_{d,subs,comp} = C_{d,subs,comp} \times \dot{m}_d \qquad C_{d,subs,comp} = BTF_{d,int,subs} \times (\dot{m}_{soildry,dc} \times C_{soildry,subs,comp} + \dot{m}_{dg,dc} \times C_{dg,subs,comp} + \dot{V}_{r,dc} \times C_{air,subs,comp}))$$

where:

- $C_{w,subs,comp}$ is the concentration in drinking water of substance *subs* emitted to compartment *comp* and other symbols represent a similar quantity for resp. fish, wet crops, meat, dry grass, wet grass and dairy products;
- $C_{water,subs,comp}$ is the concentration in (unpurified) surface water of substance *subs* emitted to compartment *comp*, other symbols represent a similar quantity for resp. the liquid fraction in soil, the solid fraction of the soil (= dryweight) and dryweight grass;
- $BCF_{f,water,subs}$ is the bioconcentration factor [28,29,30,31] linking a concentration of a substance in water to a concentration in fish, other symbols represent a similar quantity for wet crops (or grass)-liquid soil fraction ($BCF_{wg,soilliq,subs} = BCF_{c,soilliq,subs}$);
- $BTF_{m,int,subs}$ is the biotransfer factor which links the animals' daily intake of a substance *subs* to the concentration in wetweight meat, another symbol represent a similar quantity for dairy products-daily intake;
- other symbols are explained in Table 4.

Because

$$K_{air,subs,comp} = \frac{C_{air,subs,comp}}{\Phi_{subs,comp}} \tag{19}$$

equation (11) can now be rewritten as

$$K_{r,subs,comp} = K_{air,subs,comp} \times \dot{V}_r \tag{20}$$

and, equation (13) as

$$K_{o,subs,comp} = K_{water,subs,comp} \times (\dot{V}_w + BCF_{f,water,subs} \times \dot{m}_f) + K_{soilliq,subs,comp} \times BCF_{c,soilliq,subs} \times (\dot{m}_c + b_{dwt,wwt} \times [\dot{m}_m \times BTF_{m,int,subs} \times \dot{m}_{dg,mc} + \dot{m}_d \times BTF_{d,int,subs} \times \dot{m}_{dg,dc}]) + K_{soildry,subs,comp} \times (\dot{m}_m \times BTF_{m,int,subs} \times \dot{m}_{soildry,mc} + \dot{m}_d \times BTF_{d,int,subs} \times \dot{m}_{soildry,dc}) + K_{air,subs,comp} \times (\dot{m}_m \times BTF_{m,int,subs} \times \dot{V}_{r,mc} + \dot{m}_d \times BTF_{d,int,subs} \times \dot{V}_{r,dc}) \quad (21)$$

$K_{air,subs,comp}$, $K_{water,subs,comp}$, $K_{soilliq,subs,comp}$ and $K_{soildry,subs,comp}$ are modelling constants relating a flux of a substance *subs*, emitted to compartment *comp*, to a concentration in air resp. water or soil (concentration in soilwater resp. concentration in total dryweight soil). They are derived from the multimedia level III model of Mackay [23]. By estimating the daily intake $\dot{V}$ or $\dot{m}$ of air, drinking water, fish, crops, meat and dairy products by humans [27], the daily intake of dry soil and of dry grass by cattle, the conversion factor b and estimating BCFs and BTFs by the octanol/water coefficients K_{ow} [28,29,30,31], the $K_{r,subs,comp}$ and the $K_{o,subs,comp}$ can be calculated. This calculation procedure can also be followed for the reference substance.

4.5.2 The ecosystem exposure part

In the classification of ecotoxic substances, effects will be subdivided per compartment for the time being. As specific toxicity data for ecosystems in the sediment [32] and for exposure of ecosystems by air are lacking, these compartments are not yet considered. The classification of ecotoxic substances is thus limited to the compartments land soil (terrestrial ecosystems) and surface water (aquatic ecosystems). Thus, two exposure parts can be distinguished: one for exposure of terrestrial ecosystems and one for exposure of aquatic ecosystems. Again, the multimedia environmental model of Mackay [23] is used to calculate the exposure modelling constants for both terrestrial and aquatic exposure. However, the exposure routes towards man are not considered of course. In this case, the modelling constants $K_{soildry,subs,comp}$ and $K_{water,subs,comp}$ derived from Mackay models are applied directly for terrestrial resp. aquatic ecosystems.

4.6 THE EFFECT PART

As mentioned before, exposure of human beings or ecosystem species can result in a large number of effects. Apart from this, while human toxicity concerns the potential toxic effects of one species (human beings), ecotoxicity concerns the potential toxic effects of a large number of species. The question how to deal with these aspects in a no-effect level then arises.

Two possible methods can be distinguished. The first is to relate environmental exposure concentrations to the types of mechanisms initiated in humans and in ecosystem species, such as carcinogenity, mutagenity, decrease of reproduction capacity, etc. The second method is to relate the exposure concentrations to the first occurring adverse effect (based on the parameter measured in a specific toxicity test, e.g. growth, mortality, immune response, etc.) and to base the no-effect level on the threshold value for that effect.

The first method is not feasible because knowledge on mechanisms is far from complete, and because it would result in a great number of effect scores, which would be difficult to handle in

practice. The second method is current practice in modelling [24,33]. For these reasons, it is proposed here to base the no-effect levels on a threshold value for the first occurring adverse effect.

4.6.1 The human effect part

For human toxicity, two no-effect levels are distinguished: for oral effects and for respiratory effects. Since the exposure takes place by intake, the NEL should be formulated as a no-effect intake (NEI). As NEI-values for the oral no- effect level for human toxicity, the so-called *acceptable daily intake* (ADI) values and the so-called *tolerable daily intake* (TDI) values can be applied. ADIs are determined by the *World Health Organization* (WHO) for a limited number of substances. TDIs have been derived by the Dutch *National Institute for Public Health and Environmental Protection* (RIVM) for a number of priority substances for the soil sanitation [34]. For some of these priority substances an ADI was already defined by the WHO. Then the TDI is equal to the ADI. If an ADI is lacking for a priority substance, a TDI-value has been derived from comprehensive toxicological literature research.

ADIs and TDIs, can only be defined for effects that occur at doses above a certain threshold value. Carcinogenity and genotoxicity are effects for which such threshold values cannot be defined. Any dose is considered hazardous. However, by defining tolerable risk levels, e.g. an increased risk on cancer of 10^{-4} [34], so-called *virtually safe concentrations* (VSCs) have been derived. These VSCs are the basis for TDIs for carcinogenic or genotoxic substances [34].

As NEI-values for respiratory intake the so-called *tolerable air concentration* (TAC) developed by the RIVM [34] can be applied. Daily exposure to contaminated air up to the TAC-value is assumed to be equal in effect on human health to a daily oral dose the size of the TDI-value. The *air quality guidelines* (AQG) of the WHO [35] can be applied as $NEI_{r,subs}$ for substances for which no TAC has been defined yet. In both cases, a conversion is required: since the modelling constants for the respiratory and the oral route have the same dimension, the NEI-values should be made comparable too. The TAC and the AQG can be transformed into an TDI- or ADI-like intake by a conversion with the daily respiratory volume $\dot{V}_r$ and the average body weight M:

$$NEI_{r,subs} = \frac{\dot{V}_r}{M} \times (TAC \text{ or } AQG) \qquad (22)$$

If for a substance both the TAC and the AQG are lacking, the TDI or ADI can be used directly. In that case, it is thus assumed that the effect of a substance is independent from the exposure route (respiratory or oral). The hierarchy of toxicity data that can be applied as $NEI_{o,subs}$ and $NEI_{r,subs}$, is summarized in Table 1. The minimal toxicity entry needed is the TDI or the ADI.

4.6.2 The ecosystem effect part

For terrestrial and aquatic ecotoxicity, the no-effect level of a toxic substance is defined as the no (adverse) effect concentration (NEC) of the substance considered for terrestrial resp. aquatic ecosystems.

The derivation of the $NEC_{t,subs}$ and the $NEC_{a,subs}$ poses the problem that a NEC has to be derived

Table 1: Hierachy of toxicity data that can be applied as NEI for the human effect part.

hierarchy	toxicity data for $NEI_{o,subs}$	toxicity data for $NEI_{r,subs}$
1	TDI or ADI	$\dot{V}_i/M \times$ TAC
2	—	$\dot{V}_i/M \times$ AQG
3	—	TDI or ADI

for all relevant species of the ecosystem considered. The current way of determining the $NEC_{i,subs}$ and the $NEC_{a,subs}$ is to extrapolate them from single species toxicity data, such as the *lethal concentration for 50% of the organisms* (LC_{50}), the *effect concentration for 50% of the organisms* (EC_{50}) and the *no observed effect concentration* (NOEC). To this end, several extrapolation methods have been proposed [36]. The US *Environmental Protection Agency* (EPA) proposed a set of - quite arbitrary - extrapolation factors for deriving ecosystem values from single species toxicity data, taking into account the lack of data and the variance among species and assuming that 95% of all species of the ecosystem will be protected by this value [37]. The Dutch RIVM proposed some modifications for the original EPA-concept [38]. This modified EPA-method, just like the original EPA-method, estimates an "environmental concern level", which can be applied as $NEC_{i,subs}$ and $NEC_{a,subs}$. The method assumes that the ratios between acute and chronic toxicity and between laboratory single species toxicity data and field ecosystem effects are constant. The extrapolation factors used depend on the availability of single species data of members of particular taxonomic groups.

Table 2: Extrapolation factors to derive "environmental concern levels", which can be applied as a $NEC_{i,subs}$.

available information	extrapolation factor
lowest acute LC_{50}, EC_{50} or QSAR estimate of acute toxicity	0.001
lowest acute LC_{50}, EC_{50} or QSAR-estimate of acute toxicity for at least one representative of microbe-mediated processes, one representative of earthworms or arthropods and one representative of plants	0.01*
lowest chronic NOEC or QSAR-estimate of chronic toxicity for at least one representative of microbe-mediated processes, one representative of earthworms or arthropods and one representative of plants	0.1*

* Lowest value is selected in case LC_{50}S, EC_{50}, NOECs or QSARs are not available for a representative of all three taxonomic groups.

The extrapolation factors related to the available information to derive "environmental concern levels", which can be used as $NEC_{i,subs}$, are given in Table 2. The extrapolation factors used to derive "environmental concern levels", which can be used as $NEC_{a,subs}$, are given in Table 3 [38].

Van Straalen and Denneman [39] proposed a method defining so-called HC_p-values (*hazardous concentration for p% of the species*) for ecosystems based on a method developed by

Kooijman [40]. Some modifications of the method were proposed by Aldenberg and Slob [41]. The basis for the calculation of such a HC_p-value for a substance and an ecosystem are a number of (at least four) NOEC-values for characteristic and/or sensitive species. The HC_p-value for ecosystems is then calculated, based on these single species values, in a statistical way assuming a log-logistic distribution of NOEC-values for different species and protecting 95% (p=5) of the species. Wagner and Løkke developed a similar approach based on a log-normal distribution of NOEC-values for different species [42].

Table 3: Extrapolation factors to derive "environmental concern levels", which can be applied as a $NEC_{a,subs}$.

available information	extrapolation factor
lowest acute LC_{50}, EC_{50} or QSAR estimate of acute toxicity	0.001
lowest acute LC_{50}, EC_{50} or QSAR-estimate of acute toxicity for at least one representative of algae, one representative of crustaceans and one representative of fish	0.01*
lowest chronic NOEC or QSAR-estimate of chronic toxicity for at least one representative of algae, one representative of crustaceans and one representative of fish**	0.1*

* Lowest value is selected in case LC_{50}, EC_{50}, NOECs or QSARs are not available for a representative of all three taxonomic groups.

** MicroTox data may be used.

These more sophisticated extrapolation methods cannot be applied on a large scale for the time being, because for quite a number of substances toxicity data are not sufficiently available. At this moment, the EPA-approach can be applied to any substance of which at least one LC_{50}, EC_{50}, NOEC or a QSAR-estimation of one of these toxicity data is known. For reasons of comparibility, it would not be appropriate to mix the EPA-approach with the more sophisticated extrapolation methods. Hence, it is proposed to apply the practical, though quite arbitrary, EPA extrapolation method for the time being.

4.7 THE CLASSIFICATION FACTOR

Combination of the exposure part and the effect part yields the classification factor. The classification factor for human toxicity is called the *human toxicity potential* (HTP). For an emission of substance *subs* to compartment *comp* it is defined as

$$HTP_{subs,comp} = \frac{K_{r,subs,comp} / NEI_{r,subs} + K_{o,subs,comp} / NEI_{o,subs}}{K_{r,refsubs,refcomp} / NEI_{r,refsubs} + K_{o,refsubs,refcomp} / NEI_{o,refsubs}} \quad (23)$$

For the exposure parts (the different *K*s) equation (20) and (21) are elaborated according to the model description given above for both the substance studied and the reference substance. For the

effect parts (the different *NEI*s) Table 1 is used. The potential human toxic effect of the emission of a substance *subs* to compartment *comp* can now be expressed as an emission m_h of a reference substance to a reference compartment with an equivalent effect:

$$m_h = HTP_{subs,comp} \times m_{subs,comp} \tag{24}$$

The classification factor for terrestrial ecotoxicity is called the *terrestrial ecotoxicity potential* (TETP) and is defined as

$$TETP_{subs,comp} = \frac{K_{soildry,subs,comp} / NEC_{t,subs}}{K_{soildry,refsubs,refcomp} / NEC_{t,refsubs}} \tag{25}$$

where Table 2 is used for the no-effect concentration. The potential terrestrial ecotoxic effect of the emission of a substance *subs* to compartment *comp* can now be expressed as an emission m_t of a reference substance to a reference compartment with an equivalent effect:

$$m_t = TETP_{subs,comp} \times m_{subs,comp} \tag{26}$$

The *aquatic ecotoxicity potential* (AETP) is defined in a similar way:

$$AETP_{subs,comp} = \frac{K_{water,subs,comp} / NEC_{a,subs}}{K_{water,refsubs,refcomp} / NEC_{a,refsubs}} \tag{27}$$

Here Table 3 is used for the determination of the NECs. The potential aquatic ecotoxic effect of the emission of a substance *subs* to compartment *comp* can now be expressed as an emission m_a of a reference substance to a reference compartment with an equivalent effect:

$$m_a = AETP_{subs,comp} \times m_{subs,comp} \tag{28}$$

The HTP, TETP and AETP have the following properties:

- they are dimensionless;
- they are equal to one for the reference substance emitted to the reference compartment;
- they are higher for a more toxic substance and (almost) zero for a harmless substance;
- they are higher for a more persistent substance, and almost zero for a highly degradable substance.

4.8 EXAMPLE

The method described above is a theory which may be used to assess the potential toxic effects of emissions of chemicals. It is not a model to predict empirical phenomena which can be validated, such as concentrations. The particular part of the method based on the multimedia environmental models has been validated by Mackay *et al.* [43,44] as far as possible. Predicted and observed environmental concentrations appeared to be in a reasonable range.

To show what the results of the theoretical model as proposed above might look like and to show that the theoretical model is feasible in practice, we calculated the HTPs, TETPs and AETPs of phenol (chosen as the reference substance) and benzene. Air was chosen as reference compart-

ment. We emphasize that the primary aim of this example is to illustrate a new method for the assessment of emissions of toxic chemicals within the framework of LCA and not to give the ultimate HTP, TETP- and AETP-values for these substances. We selected these two substances for practical reasons, as data for these substances are quite well available and they fit into the fugacity approach without any further adaptations (see discussion).

The level III model and data applied to calculate the environmental concentrations are largely based on a model described by Mackay & Paterson [44]. Additional data needed to calculate the HTPS, TETPS and AETPS of phenol and benzene are shown in Table 4. The values are reported by Toet *et al.* [24] and De Nijs & Vermeire [27]. A comprehensive listing of the model applied here is available from the authors on request.

Table 4: Input data for the calculation of HTPS, TETPS and AETPS [24, 27].

symbol	quantity	value	unit
M	human mass	70.0	kg
$b_{dwt,wwt}$	factor for conversion of wet plant weight to dry plant weight	4.0	—
$\dot{m}_{dg,dc}$	dairy cattle daily dryweight grass consumption	16.9	$kg \times day^{-1}$
$\dot{m}_{soildry,dc}$	dairy cattle daily dryweight soil consumption	0.41	$kg \times day^{-1}$
$\dot{V}_{r,dc}$	dairy cattle daily respiratory volume	122	$m^3 \times day^{-1}$
$\dot{m}_{dg,mc}$	meat cattle daily dryweight grass consumption	12.2	$kg \times day^{-1}$
$\dot{m}_{soildry,mc}$	meat cattle daily dryweight soil consumption	0.39	$kg \times day^{-1}$
$\dot{V}_{r,mc}$	meat cattle daily respiratory volume	122	$m^3 \times day^{-1}$
$\dot{m}_f$	human daily fish consumption	0.01	$kg \times day^{-1}$
$\dot{m}_c$	human daily wetweight crops consumption	0.558	$kg \times day^{-1}$
$\dot{m}_m$	human daily meat consumption	0.126	$kg \times day^{-1}$
$\dot{m}_d$	human daily dairy products consumption	0.371	$kg \times day^{-1}$
$\dot{V}_w$	human daily drinking water volume	2.0	$l \times day^{-1}$
$\dot{V}_r$	human daily respiratory volume	20.0	$m^3 \times day^{-1}$

The physical properties and toxicity data of phenol and benzene needed as input for a HTP, TETP and AETP calculation with a Mackay Level III model are shown in Table 5 along with their values. The values are from Mackay [23], Mackay & Paterson [44], Howard [45], Vermeire *et al.* [34], Stortelder *et al.* [46] and Denneman & van Gestel [47].

Table 5: The physical properties and toxicity data of phenol and benzene.

quantity	unit	phenol	benzene
molecular weight	$g \times mol^{-1}$	94.1 [31]	78.1 [31]
vapor pressure at 25°C	Pa	70.6 [31]	1.27×10^{4} [31]
water solubility	$g \times m^{-3}$	8.20×10^{4} [31]	1.78×10^{5} [31]
^{10}log octanol-water coefficient	—	1.46 [31]	2.13 [31]
melting point	°C	40.9 [31]	5.53 [31]
degradation rate constant air	h^{-1}	4.62×10^{-2} [45]	8.60×10^{-4} [44]
degradation rate constant water	h^{-1}	2.17×10^{-2} [45]	4.80×10^{-3} [44]
degradation rate constant soil	h^{-1}	2.27×10^{-1} [45]	0
degradation rate constant sediment	h^{-1}	0	0
$NEC_{r,subs}$	$kg \times m^{-3}$	2.10×10^{-7} [34]	3.00×10^{-8} [34]
$NEI_{o,subs}$	$kg \times kg^{-1} \times day^{-1}$	6.00×10^{-8} [34]	4.30×10^{-7} [34]
$NEC_{i,subs}$ at an organic carbon content of 2%	$g \times kg^{-1}$	6.42×10^{-5} [47]	1.00×10^{-11} [47]*
$NEC_{a,subs}$	$g \times l^{-1}$	1.70×10^{-5} [46]	3.50×10^{-5} [46]

* If a NEC value is unknown it is assumed to be 1.00×10^{-11} (see discussion).

The results of the calculations are shown in Tables 6 and 7. The toxicity potentials of benzene are dominated by its high volatility, which is seen in relatively high values for the *K/NEI* ratio for the respiratory route. Furthermore, it is shown that the AETP and the TETP are the highest for emissions to resp. air and water, which is explicable as aquatic ecosystems will be most affected by direct emissions to water and terrestrial ecosystems will be most affected by direct emissions to soil. Notice that the high values for the TETP of benzene are caused by the 10^{-11}, due to the absence of a terrestrial NEC for benzene.

Table 6: The HTP of phenol and benzene; phenol and air are taken as *refsubs* resp. *refcomp*.

	phenol			benzene		
initial emission compartment	K_i/NEI_r	K_o/NEI_o	*HTP*	K_i/NEI_r	K_o/NEI_o	*HTP*
air	1.72×10^{-5}	8.43×10^{-5}	1	6.57×10^{-3}	2.11×10^{-5}	64.9
water	1.55×10^{-9}	1.13×10^{-5}	0.111	7.02×10^{-4}	2.32×10^{-5}	7.14
soil	6.81×10^{-9}	4.32×10^{-3}	42.6	6.43×10^{-3}	4.01×10^{-3}	103

Table 7: The TETP and AETP of phenol and benzene; phenol and air are taken as *refsubs* resp. *refcomp*.

type of potential & initial emission compartment	phenol		benzene	
terrestrial ecotoxicity	*K/NEC*	*TETP*	*K/NEC*	*TETP*
air	9.2×10^{-11}	1	1.75×10^{-3}	1.91×10^{7}
water	8.28×10^{-15}	9.01×10^{-5}	1.88×10^{-4}	2.04×10^{6}
soil	4.73×10^{-9}	51.5	3.43×10^{-1}	3.73×10^{9}
aquatic ecotoxicity	*K/NEC*	*AETP*	*K/NEC*	*AETP*
air	1.67×10^{-8}	1	3.85×10^{-8}	2.30
water	6.78×10^{-7}	40.5	1.33×10^{-6}	79.6
soil	1.98×10^{-9}	0.118	5.28×10^{-8}	3.15

4.9 DISCUSSION

The method for the classification of toxic substances discussed above needs further elaboration. Aspects to be considered include the choice of the reference substance, the adaptation of Mackay models for different groups of substances, the data needed and refinement of the human exposure routes in the model.

In the selection of the reference substance technical criteria are relevant. Because the modelling constant of the reference substance $K_{refsubs,refcomp}$ is in the denominator of the definition of the classification factor, the most important technical criterium is that the modelling constant of the reference substance be non-zero. This means that in an equilibrium situation, an emission of the reference substance to air results in a concentration both in soil and in water in order to be able to calculate the exposure part of the classification factors for ecotoxicity. Another important technical criterium is that for the reference substance an ADI- or TDI-value is available. Phenol seems to meet these criteria as it is a substance that is dispersed through all environmental compartments and a TDI-value is available; a TAC- or AQG is lacking and a NEC_r has to be derived from the TDI according to equation (22). Of course, it is possible to choose another reference substance in the model proposed. If a reference substance of which many toxicological data are known is chosen, it may be interesting to investigate the possibility to use data from comparable ecotoxicological experiments instead of applying the EPA extrapolation factors. Moreover, the use of extrapolation factors is disputed; for example, it does not consider bioaccumulation through foodchains [48].

The Mackay model as discussed above uses fugacity as an equilibrium criterion to determine the environmental fate of emissions of chemicals. This approach is suitable for chemicals which can establish measurable concentrations in the vapor phase. It is not applicable to some metals, organometals, ionic compounds and some organics such as polymers that lack a vapor pressure [49]. Mackay & Diamond [49] proposed to use a so-called "equivalent aqeous" concentration instead of the fugacity as an equilibrium criterion for substances that lack a vapor pressure. In this case, however, empirical data about the magnitude of the partitioning coefficients between various environmental media of a particular substance are necessary. As these are often lacking, this problem clearly needs further attention.

To calculate the different modelling constants based on a Mackay level III model, quite a number of physical data and toxicity data per substance are required, see Table 5. As mentioned above, the type of physical data required may be different for inorganic substances. Some of these data are documented in comprehensive handbooks [45,50], but it will be difficult to gather all these data for the most relevant toxic substances, especially data on degradation kinetics. As the practical applicability of the HTP-, TETP- and AETP-approach depends on the availability of these data, it should be emphasized that these data should preferably be gathered in a more structural way and stored in a database, and be regularly updated to the latest state of knowledge. Despite the fact that some data may be missing for the time being, we still think that this approach is a substantial improvement of former methods for the classification of toxic chemicals, because it includes exposure routes and relevant toxicity data. To deal with this current lack of data it is suggested that if for a chemical degradation data or toxicity data are missing, they are assumed to be zero until proven untrue. For computational reasons, a lacking toxicity entry should be approached by a very small number instead of 0, e.g. 10^{-11}.

The modelling of the exposure routes to human beings might need further improvement. Basically, an emission of a substance can lead to human exposure in three ways:

- direct: via respiration and consumption of (drinking) water;
- indirect: via the consumption of fish, meat, dairy products and crops;
- secondary indirect: by consumption of the same products produced with sludge and manure, which are polluted "co-products" of drinking water purification resp. meat production.

In the model described, the direct and indirect routes are included, although incompletely. For example, exposure by skin contact (soil, air, water or the product itself) is not included. Two other examples are the exposure to a substance by drinking water and respiration. In equation (14) the concentration of the substance in drinking water is assumed to be equal to the concentration in unpurified surface water, thus not considering drinking water purification systems [cf. 24]. The exposure to a substance by respiration (equation (12)) includes a similar "overestimation" as a part of the world population respires mainly relatively "airconditioner clean" indoor air. These obvious "over-estimation" of human exposure by drinking water and respiration can be compensated by including averaged purification data per substance, if known. If such data are included, it is important to also include the secondary indirect routes (which are not yet considered at all). Secondary exposure routes exist if the exposure routes pass economic processes, such as agriculture and purification of drinking water. Modelling of these processes raises some problems. Economic processes have a number of inputs, such as grass and cattle feed resp. unpurified surface water, and a number of outputs, such as meat, dairy products and manure resp. drinking water and sludge. Exclusion of this secondary indirect exposure route is probably justified if the degradation time of a substance in sludge and manure that is used again as a fertilizer in agriculture, is relatively low compared to the time it takes to pass the secondary route. However, for persistent substances such as heavy metals and some pesticides, exclusion seems not justified. The extent of this potential "underestimation" can be calculated by, for example, estimating the amount of the substance considered in the manure. This amount could then be defined as an emission again and the secondary indirect exposure could be calculated with the same model. It is recommended here, that the influence of this secondary indirect exposure be further investigated. Refinement of the direct and indirect routes together with the inclusion of secondary indirect routes are subjects for further research.

Finally, it is suggested to create an international scientific panel, which could discuss proposals for classification factors, such as these here, and coordinate scientific efforts made in the different environmental fields. Comparable with the *scientific assessment panel* for ozone depletion under the auspices of the *World Meteorological Organization* (WMO) and the *intergovernmental panel on climate change* (IPCC) for global warming under the auspices of WMO and *United Nations Environment Programme* (UNEP), panels might be established for human toxicity and ecotoxicity. With respect to the latter, initiatives are currently being taken.

4.10 NOMENCLATURE

ADI acceptable daily intake ($kg \times kg^{-1} \times day^{-1}$)

AETP aquatic ecotoxicity potential (dimensionless)

b	factor for the conversion of wet grass weight into dry grass weight
BCF	bioconcentration factor ($m^3 \times kg^{-1}$, $kg \times m^{-3}$ or $kg \times kg^{-1}$)
BTF	biotransfer factor ($day \times kg^{-1}$)
C	concentration ($kg \times m^{-3}$ or $kg \times kg^{-1}$)
EC_{50}	effect concentration for 50% of the organisms ($kg \times m^{-3}$ or $kg \times kg^{-1}$)
K	modelling constant; the ratio between the human daily exposure for human beings or equilibrium exposure concentration for ecosystems and the emission flux of a substance (dimensionless, $day \times kg^{-1}$ or $day \times m^{-3}$)
HTP	human toxicity potential (dimensionless)
I	daily intake of a substance ($kg \times day^{-1}$)
L	classification factor (dimensionless)
LC_{50}	lethal concentration for 50% of the organisms ($kg \times m^{-3}$ or $kg \times kg^{-1}$)
m	emission-pulse, dose (kg)
$\dot{m}$	mass of food consumed per day ($kg \times day^{-1}$)
M	average human body weight (kg)
NEC	no (adverse) effect concentration ($kg \times m^{-3}$ or $kg \times kg^{-1}$)
$NOEC$	no observed effect concentration ($kg \times m^{-3}$ or $kg \times kg^{-1}$)
NEI	no-effect intake ($kg \times kg^{-1} \times day^{-1}$)
NEL	no-effect level ($kg \times kg^{-1} \times day^{-1}$, $kg \times m^{-3}$ or $kg \times kg^{-1}$)
t	time (day)
T	toxic effect (kg or dimensionless)
TDI	tolerable daily intake ($kg \times kg^{-1} \times day^{-1}$)
$TETP$	terrestrial ecotoxicity potential (dimensionless)
$\dot{V}$	volume consumed per day ($m^3 \times day^{-1}$)
VSC	virtually safe concentration ($kg \times m^{-3}$ or $kg \times kg^{-1}$)
Φ	emission-flux ($kg \times day^{-1}$)

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4.13 EPILOGUE*

[...] It is impossible to base statements on toxicity in LCA on concentrations below the standard, be it MAC, NOEC or ADI. This is a difficult point, which will be worked out below in some more detail.

Assume that two methods of shaving are compared: the functional unit is one shaving activity, method A is with a razor blade and method B involves an electric razor. The life cycle of method A includes the production of shaving-soap. Assume this takes place in a small factory. The life cycle of method B includes the production of PVC, which will be assumed to take place in a large plant. A result of the analysis might be that method A including the production of shaving-soap needed for one shaving is environmentally worse than method B including the production of PVC needed for one shaving.

However, due to the large production volume of the PVC plant, the PVC process *in its actual extent* is worse than the shaving-soap process *in its actual extent*. This aspect cannot be considered by LCA. LCA emission data are obtained by dividing yearly emission amounts by the yearly production amounts. The operating *time* of the process is then divided out and the result is a number of emission loadings per amount of product produced. In LCA the volume of a specific process has thus become irrelevant. The process in its actual extent is only relevant for Environmental Impact Assessment (EIA) and Risk Assessment (RA). This makes that with an LCA, statements in terms of actual risks cannot be made. Even if you would like to do so, just forget it.

We hope to have demonstrated above that it is fundamentally impossible to perform an actual risk analysis within the framework of LCA. But what do these results represent? The results of a life cycle impact assessment do not represent *actual* risks but *potential* risks. No one will die of the emissions for one shaving. But all tiny contributions of all activities make together the environmental problem.

LCA is not concerned with the degree to which a NOEC is actually *exceeded*, but with the degree to which it is potentially *filled up*. We still believe that the NOEC can be used as a suitable measure for the strength of a toxic substance in LCA. The exact form of the dose-response curve is essential for an *actual risk assessment*. Of course, the actual impacts are important as well. Actual assessments, such as RA and EIA, may thus never be superseded by LCA. [...]

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* Quoted from "Heijungs, R. and J.B. Guinée, 1993: *CML on actual versus potential risks*. In: SETAC Europe News (LCA News), Vol. 3, No. 4, p. 4."

Chapter 5

A proposal for the definition of resource equivalency factors for use in product Life-Cycle Assessment

Reprinted with permission from *Environmental Toxicology and Chemistry*, Volume 14(6), J.B. Guinée and R. Heijungs, "A Proposal for the Definition of Resource Equivalency Factors for Use in Product Life-Cycle Assessment." Copyright © 1995 SETAC. Appendices 2, 3 and 4 are new materials.

5.1 ABSTRACT

Environmental Life-Cycle Assessment of products (LCA) has been the focus of growing attention in the last few years. The methodological framework has been developed rapidly, and a provisional "Code of Practice"' has been drawn up by an international group of experts. One of the elements of LCA is impact assessment, which includes a characterization step in which the contributions of resource extraction and polluting emissions to impact categories such as resource depletion, global warming and acidification are quantified and aggregated as far as possible. This can be achieved by multiplying extractions and emissions by a so-called equivalency factor and aggregating the results in one or more effect score(s) per impact category.

In this paper a proposal is developed for equivalency factors indicating the relative depletion of a resource per unit extracted. It is proposed to measure depletion by physical data on reserves, production and regeneration rates and to distinguish between abiotic and biotic resources. Equations are developed to calculate equivalency factors for these two categories of resources, resulting in so-called *abiotic depletion potentials* (ADP) and *biotic depletion potentials* (BDP). ADP- and BDP-values are provided for a number of resources and the application of these ADPs and BDPs in LCA is illustrated.

5.2 INTRODUCTION

Environmental Life-Cycle Assessment of products (LCA) is a rapidly developing area of applied environmental science. LCA is a quantitative tool for assessing the environmental impacts of functions and services delivered by product systems. The term *product system* is understood to mean the product during its entire life-cycle, from the cradle to the grave, in terms of all the economic processes concerned. The term economic process refers to any kind of process producing an economically valuable material, component or product or yielding an economically valuable service (transport or waste management, for example). Environmental impacts are allocated to all the materials, components and/or products produced.

Since 1990 there has been an enormous growth in the number of LCA studies. Several comprehensive methodological projects have been undertaken and an initial version of a "Code of Practice" has been drafted by an international committee of LCA-experts [1]. One of the main elements of this "Code of Practice" is a methodological framework comprising four components: goal definition and scoping, inventory analysis, impact assessment and improvement assessment.

Goal definition and scoping define the subject of study and the *functional unit* to be investigated. In the *inventory analysis* all extractions of resources and emissions of hazardous substances attributable to the product system(s) studied are listed in an *inventory table*. In the *impact assessment* it is first determined which impact categories (environmental problem types) are to be considered and which extractions and emissions contribute to which of these impact categories. In the "Code of Practice" this step is called *classification*. In a following step the *characterization*, the "analysis/quantification, and where possible, aggregation of the impacts within the given impact categories" takes place [1]. This can be achieved by multiplying extractions and emissions by an equivalency factor and by aggregating the results of these multiplication

in one *effect score* for each impact category:

$$effect\ score_{problem\ type} = \sum_i equivalency\ factor_{i,problem\ type} \times emission_i \text{ or } extraction_i \qquad (1)$$

The equivalency factor is defined for each type of resource or emission and for each impact category (resource depletion, global warming, acidification, etc). To permit better interpretation, it has been proposed to carry out normalization of the effect scores [1,2,3,4]. To this end, the effect scores of a particular product (system) are divided by the total extent of the relevant effect for a certain area (e.g. the world) and a certain period of time (e.g. one year). An initial attempt to calculate so-called world annual effect scores has been made by Guinée [5]. The final step of impact assessment is *valuation*, in which the relative importance of each of the impact categories is assessed. Valuation results may be used as a basis for choosing among product alternatives, or for the purposes of product improvement. The last component of the "Code of Practice" framework is *improvement assessment*, in which the options for improving the product system(s) under study are identified. For a more comprehensive discussion of the principles and elaboration of the individual steps, we refer to previous publications on LCA-methodology [2,3,6,7,8,9,10].

One of the elements of LCA that has received extensive attention in the past year is characterization [11,12,13]. Equivalency factors have been suggested for the following impact categories: *global warming potentials* (GWPs) [14], *ozone depletion potentials* (ODPs) [15], *photochemical ozone creation potentials* (POCPs) [16], *acidification potentials* (APs) [2,17], and *nutrification potentials* (NPs) [2]. Moreover, Guinée and Heijungs [18,19] have proposed developing similar potentials - which have been challenged by Assies [20] and defended by Heijungs and Guinée [21] - for human-toxic and ecotoxic emissions.

For the depletion of resources a full set of equivalency factors is still lacking. Heijungs *et al.* [2] and Fava *et al.* [22] have begun to explore this area, but their work was provisional. In this paper we elaborate further on this topic. The core issue addressed is: how serious is the depletion of one particular natural resource in relation to that of another, and how can this be expressed in terms of equivalency factors for these resources. A typical question which arises in LCA is the following. A certain function (e.g. packaging of beverages) can be fulfilled by two alternative product systems (e.g. aluminium can or glass bottle). How do these alternatives compare with respect to depletion?

In this paper, first the concept of resource depletion is defined. This leads to a discussion of the parameters relevant for measuring depletion, and how these parameters are to be combined to arrive at an appropriate equation. The paper ends with an illustration of the application of the proposed approach and some general conclusions.

5.3 RESOURCE DEPLETION AS AN ENVIRONMENTAL PROBLEM

Environmental problems can be defined as mechanisms or impacts harming human health, biodiversity and material welfare [23,24]. For the depletion of abiotic resources, the material welfare aspect is particularly important. Depletion of resources, or scarcity, as an environmental

problem, then, is the decreasing availability of natural resources influencing one or more of these three societal value fields.

Availability is not taken to mean the availability of a resource in the economy. Reuse of aluminium from the economic stock, e.g. from discarded window frames, aluminium foil or aluminium cans, is not considered a depletion problem. Even after excluding the economic stocks, in this definition the term availability is still an ambiguous one. It can mean at least two things: availability in the primary extraction medium (e.g. iron available from iron ore in the earth's crust) or availability in the entire geosphere and biosphere (thus including iron available from landfills, plants, animals, (sea)water, soil, atmosphere, etc.). If availability is defined as availability in the primary extraction medium (narrow sense), all types of resources can in principle be depleted if annual extraction (= production) exceeds annual regeneration. If availability is defined as availability in the geosphere and biosphere (broad sense), however, elemental materials such as iron can never be depleted unless they are emitted into outer space or transmuted in nuclear processes.

The precise definition of the term availability determines which resource types are relevant in terms of depletion. For instance, when the availability of mineral resources is defined in a broad sense, the main problem is the increasing amount of energy consumed and solid waste produced in extracting ever lower concentrations of the mineral in question [25], or in recovering already extracted but dispersed resources. An example of such a dispersed resource is lead. Lead is extracted from lead ore confined in a mine, and then dispersed in the environment, e.g. as an emission due to leaded gasoline combustion. In principle, this lead can be recovered, but the concomitant problems are virtually unsurmountable, involving ever more extraction of fossil fuels and ever greater emissions of CO_2, SO_2, NO_x, etc. In terms of depletion, then, the main problem of mineral utilization *might* be a growing demand for energy resources and, in some cases, the lack of substitutes. Huppes [24] argues that depletion of energy resources may not be that problematical because of the still growing known reserves of fossil fuels and the major potential of cleaner substitutes such as solar energy. However, development of sustainable and clean energy sources and closing of substance cycles, preventing emissions of CO_2, SO_2, NO_x, etc., will be a long-term process and will probably never succeed completely. It seems appropriate, therefore, to take a conservative approach, for the time being applying a narrow definition of availability (i.e. stocks in the primary extraction media) and considering how the approach might be adapted if the broad definition of availability becomes more applicable in the future.

The term natural resources also needs clarification. By natural resources we mean the minerals and materials to be found in the earth, sea or atmosphere and biota which have not yet been industrially processed, e.g. iron ore rather than iron, bauxite rather than aluminium and elephants rather than ivory.

The economic process of resource extraction generally has a number of environmental impacts, such as release of chemicals, generation of waste flows, consumption of fuels, etc. Another type of impact is damage to ecosystems and landscapes resulting from mining activities. These are all accounted for by including the process of extraction in the product life cycle and distinguishing various impact categories such as depletion, global warming, ozone depletion, acidification and land use (including ecosystem and landscape degradation). This paper focuses on the elaboration of equivalency factors for depletion.

Because of the intrinsic value of biotic resources, their source function and their role in the maintenance of the life support system, we propose to assess the depletion of biotic resources separately from that of abiotic resources. The valuation step of impact assessment provides an opportunity to assign a different significance to biotic depletion.

Another commonly made distinction is between renewable and non-renewable resources. The renewability criterion is applicable to biotic resources (tropical tree growth, elephant reproduction, etc.) and, on a mainly geological time scale, also to abiotic resources such as fossil fuels. In our opinion, renewability should be accounted for in the elaboration of equivalency factors for depletion rather than being used as a criterion for categorizing resource types. The situation is comparable with the distinction between degradable and non-degradable toxic substances. It makes no sense to define separate effect scores for the toxic effects caused by these two categories of chemicals. Instead, degradability should be incorporated in the equivalency factor [18,20].

Abiotic resources include all fossil and mineral resources. They should be quantified in terms of elements or compounds, depending on the form relevant for depletion. With aluminium, for example, it is the element aluminium in the ore (e.g. bauxite) that matters and not the specific chemical composition of the ore (e.g. Al_2O_3 or Al_2SiO_5). With fossil fuels, it is the specific chemical compound that is convertible to energy that matters and not the constituent elements (e.g. natural gas rather than the elements carbon and hydrogen). Although consideration might be given to quantifying fossil fuels in terms of their energy content, since fuels are not directly interchangeable (e.g. coal cannot be used as a car fuel) and *some* fossil fuels can be used to produce materials (e.g. crude oil for plastics) instead of energy, it is proposed here to treat the various fossil fuels similarly to individual elements (such as aluminium).

The scope should be limited to depletable elements and compounds. Elements such as oxygen, hydrogen, carbon and nitrogen are thus excluded and only a few specific compounds of these elements (fossil fuels, natural diamond) are considered. Furthermore, elements that do not occur naturally but can only be artificially produced by nuclear reactions, such as americium, astatine, etc., are excluded. The same applies to artificial isotopes such as tritium, ^{60}Co, ^{138}Ba, etc.

With respect to biotic resources, all resources which are extracted by man should be included, e.g. elephants (billiard balls, although already banned), crocodiles (bags), fishes (consumption), etc. However, biotic resources are considered only in so far as they are not reproduced by a production process: the harvesting of wood from a forest plantation, for example, is not considered to be an activity inducing biotic depletion problems, but a production process within the boundaries of the product system, with its specific environmental impacts (e.g. from the use of fertilizers, pesticides and energy).

Finally, an important distinction can be made between stock resources and flow resources. This paper considers only stock resources, i.e. resources with a limited available stock, such as iron ore and elephants. The use of flow resources, such as solar energy and cooling water, is a different matter. For these resources there is no stock to be depleted, but rather a maximum utilizable flow. It is still an open question whether and how an effect score for the use of flow resources can or should be designed and, if so, whether it can be integrated with the effect score for stock resources.

5.4 CONCEPTS FOR ASSESSING DEPLETION

The seriousness of the depletion of a given resource is determined by a number of factors. Obviously, the amount available (the reserve) is important. Another parameter is the annual deaccumulation rate, i.e. production (e.g. in kg/yr) minus regeneration (same unit). Production of a resource is defined here as the total amount of the resource extracted annually, while regeneration is the total amount created annually. The prosperity created by using the resource is also important, as is the possibility for substitution by other resources. If the seriousness of depletion were assumed to be equal for all resources, the resources could be aggregated in mass or volume terms without any further assessment. However, mass and volume are not indicators of the differences in abundance, social value, etc. among resources. In our opinion, these latter parameters should be part of a measure of the seriousness of depletion and we therefore reject this "aggregation without any further assessment" method. On this basis, then, two concepts can be distinguished to measure the seriousness of the depletion of a resource: depletion measured by economic data (e.g. price) and depletion measured by physical data (e.g. reserves, deaccumulation, entropy).

5.4.1 Depletion measured by price

Although the price of a resource can be considered as a measure of its level of depletion or scarcity and its social value, it reflects more than these two aspects. The structure of particular economic markets, national social conditions reflected in the cost of labour, the power of mining companies with a monopoly, and the self-interest of governments eager for short-term profits all interweave to form a price that is not a good indicator of present and future scarcity [cf. 26].

5.4.2 Depletion measured by physical data about reserves and/or deaccumulation

Physical data on the four parameters of reserves, deaccumulation, substitution and prosperity can provide an indication of a resource's scarcity. While this approach is suitable for resource reserves and deaccumulation, the latter two parameters are much more difficult to express or measure in physical terms. Substitution could, for example, be expressed in terms of the time needed to substitute the original resource by a functionally equivalent one, but precise measurement remains essentially unfeasible. The search for substitutes will often only be started when it becomes clear that a particular resource is *really* becoming scarce. The social value of a resource is also difficult to express and measure in physical terms. To the extent that it is possible at all, it is reflected in the physical data on the reserves and deaccumulation rate of the resource.

In a conservative approach aimed at ensuring 'narrow sense' resource availability for future generations, we prefer to focus on physical facts (reserves and deaccumulation) and exclude value-bound parameters (substitution and social value). The possibility for substitution depends, for example, on the efforts made to investigate the elements and their potential applications, while the social value of a resource depends, for example, on the level of development of the society in question. In our opinion, these value-bound parameters should not be part of the scientific characterization but rather be treated in social debate and through value-based valuation. In this paper we therefore restrict ourselves to elaborating measurement of depletion in terms of physical data on reserves and deaccumulation.

That both these quantities need to be taken into account is illustrated by Table 1, which gives the reserves and deaccumulation rates of four hypothetical resources A, B, C and D [cf. 22].

Table 1: Reserves and deaccumulations of four hypothetical resources

resource	A	B	C	D
reserve (Gton)	1	0.0001	1	1
deaccumulation (Gton/yr)	0.000001	0.000001	0.1	0.0001

First, consider resources A and B, which differ only in the scale of their reserves. Depletion of B seems to be a more serious problem than depletion of A, i.e. using 1 kg of B is worse than using 1 kg of A. Next, consider resources C and D. Their reserves are equal and the only difference is the higher deaccumulation rate of C. Intuitively, we consider extraction of 1 kg of C more problematical than extraction of 1 kg of D. In other words, extraction of resource C should be discouraged as much as possible, and LCA should provide an incentive to prefer resource D. Another argument derives from renewable resources: the seriousness of extracting a given renewable resource is determined partly by the degree of exploitation involved. This can be accounted for by including deaccumulation (production minus regeneration).

In conclusion, both reserves and deaccumulation are important and should somehow be included in an equation for the equivalency factors indicating the seriousness of depletion.

The concept of entropy or exergy might also be used to characterize resources, as proposed by Finnveden [27]. The increase in entropy (or the loss of exergy) could be used as an indicator of the level of structure or chaos (or the ability to do work). The entropy of natural resources confined in natural ores is relatively low, while that of natural resources dispersed through the environment is high. Earlier, we argued that a broad definition of availability would be better for some abiotic resources, especially for those expressed in elemental terms (all abiotic resources except fossil fuels, diamond, etc.). In combination with appropriate data on reserves, entropy might in future serve as a good indicator for the efforts required to recover dispersed abiotic resources. However, as Finnveden concludes in his paper, the entropy/exergy approach needs further development before any conclusions can be drawn as to its applicability as a characterization method in the context of LCA [27].

5.5 DATA: CONCEPTS AND SOURCES

Before elaborating an approach for assessing depletion based on a physical measure, the required type of data on reserves and deaccumulation needs some discussion.

5.5.1 Reserve

With respect to abiotic resources, there are a number of different definitions of the term "reserve of resources". The U.S. Bureau of Mines [28] has developed nomenclature for these various definitions. Two classes from this nomenclature are discussed here (reserve base and economic reserve) and two new classes introduced (ultimate reserves and ultimately extractable reserves).

The *reserve base* is that part of an identified resource that meets specified minimum

physical and chemical criteria related to current mining practice. It encompasses that fraction of the resources with a reasonable potential for becoming economically available within planning horizons beyond those that assume proven technology and current economics [28]. The *economic reserve* is that part of the reserve base which can be economically extracted at the time of determination [28]. Economic reserves are defined as reserves which are economically attractive to explore because of high local concentrations of the resource or a good local infrastructure to transport the resource (railways, rivers etc.). All kinds of considerations may play a role in terming a reserve economically attractive, but they are unrelated to the depletion problem of that particular resource. For example, the economic reserves of most resources have increased over the last few decades, although to different extents, whereas the actual depletion problem must necessarily have increased in this period through extraction. Unfortunately this major drawback also holds for the reserve base, as technological development depends on financial investments [29]. In fact, it is not in the interest of resource-extracting companies to have too large economic reserves and reserve bases because this could lower their market prices [24].

Because of these drawbacks, we propose using another definition of reserve: *ultimate reserve*, also known as geological reserve [29] or resource base [30]. Ultimate reserves are estimated by multiplying the average concentrations of chemical elements in the earth's crust by the mass of the crust. To cover all potential primary extraction media, reserves in the oceans and atmosphere might also be added. This proposal is also to be found in a recent report by the Dutch *National Institute for Public Health and Environmental Protection* (RIVM) [31], and is very useful for comparative characterization of resources with respect to depletion. However, ultimate reserves cannot be extracted completely, as some locations will be inaccessible, e.g. reserves in the sea bed. Those reserves that can ultimately be technically extracted may be termed the *ultimately extractable reserves*. Between the ultimate reserve and the ultimately extractable reserve there is likely to be a substantial difference.

In terms of depletion, the ultimately extractable reserve is the only relevant reserve parameter. However, data on this type of reserve are unavailable and will never be exactly known because of their dependence on future technological developments. Provisionally, then, we propose to apply the ultimate reserve concept (earth's crust, oceans and atmosphere), implicitly assuming the ratio between the ultimately extractable and ultimate reserve to be equal for all resource types. The data for this concept can be derived from [32,33,34,35,36].

For fossil fuels and natural diamonds there are only proven reserve data [37], which are defined somewhat differently from reserve base data [cf. 28,37]. The ultimate reserves of fossil fuels and natural diamond cannot be estimated in the same way as those of other resources, since with these two categories it is not the basic element that matters but the specific compound. However, for fossil fuels a rough estimate can be made on the basis of the fossil carbon content of the earth's crust, which has been estimated by Berner *et al.* [38]. This C-content can be converted to ultimate reserves estimates for natural gas, crude oil and coal by calculating the total C-content of the proven reserves of fossil fuels, for which data are available [39], and multiplying the proven reserve of a fossil fuel by the ratio of the total C-content as estimated by Berner *et al.* to the total C-content of the proven fossil fuel reserves. For an explanation of these calculations we refer to appendix 2 (p. 104). For natural diamonds similar reasoning might be possible, but the appropriate data are not known to the authors.

For biotic resources these types of differences are not relevant. Biotic resources are generally present on the earth's surface and can therefore be estimated directly. However, data sources are spread around the world and therefore difficult to gather. This hampers practical elaboration of a physical method for characterizing biotic resources. Data on certain species such as the black rhinoceros, the African elephant and certain whales can be derived from [40,41]. Data on tropical hardwood reserves and deaccumulation are only available in m^2 of forest ecosystems [42] and not in kg or m^3 of hardwood as a resource. As this resource is such an important item in the environmental debate, it is included here in m^2, realizing that this implies a certain overlap with land use, which is separately assessed within LCAs (see above). As soon as tropical hardwood data become available in kg or m^3 of resource, this situation can be improved.

5.5.2 Deaccumulation

Data on deaccumulation can, if necessary, be compiled from data on production and regeneration. For abiotic resources such as fossil fuels and diamonds, production data are sufficient, assuming regeneration is negligible in relation to production.

There is reliable information available on annual world production of a wide range of abiotic resources. The U.S. Bureau of Mines [28,43] publishes such data for some eighty different types of abiotic resources; however, this does not cover the complete set of relevant abiotic resources. Production data are known for chemical compounds, while geological reserve data are available for individual elements. Production data of these compounds must be translated into data for individual elements on the basis of molecular weight, e.g. 1 kg of NaCl implies 23/58.5 kg Na and 35.5/58.5 kg Cl.

For biotic resources the relevant data are far more difficult to obtain. There is not a single institute which makes an annual inventory for any substantial number of endangered species, although IUCN has working groups for individual types of endangered habitat and species. The importance of a coordinated effort to regularly publish the relevant data should be stressed. Deaccumulation data are only available for a limited number of species, mentioned in the previous paragraph [40,41].

5.6 EQUATIONS FOR EQUIVALENCY FACTORS AND EFFECT SCORES

In this section equations are defined for calculating equivalency factors for depletion based on two parameters: reserves and deaccumulation.

One requirement should immediately be stressed: the units chosen for the equivalency factor, e.g. volume or mass, should not influence the ultimate appraisal of depletion for the product systems being compared. In other words, for any two given product systems the ratio between the effect scores based on equivalency factors in kg should not be different from that based on equivalency factors in m^3. The unit of the equivalency factor thus determines in which unit extraction should be expressed.

A general equation for calculating the effect score for depletion is given by:

$$depletion = \sum_{res} equivalency\ factor_{res} \times extraction_{res} \qquad (2)$$

One possibility for the equivalency factor is to take the reserves-deaccumulation ratio, which for abiotic resources is better known as the reserves-production (R/P) ratio or as the statistical reserve index [44]. It includes both reserves and deaccumulation, or production for abiotic resources. The R/P ratio has the dimension of time. This time unit can be interpreted as the sufficiency of the reserve: the number of years for which current reserves will suffice at the current deaccumulation or production level. An equivalency factor can now be defined as the inverse of the R/P ratio. Multiplication of this equivalency factor by the quantity of resource required to produce a functional unit of product now results in the following effect score for depletion:

$$depletion(\mathrm{kg \cdot yr^{-1}}) = \sum_{res} \frac{deaccumulation_{res}(\mathrm{kg \cdot yr^{-1}})}{reserve_{res}(\mathrm{kg})} \times extraction_{res}(\mathrm{kg}) \tag{3}$$

It can be shown, however, that this method does not meet the previously stated requirement of being invariant with respect to a change from kg to m^3 (see Appendix 1, p. 106). A more sophisticated problem is that the share in the reserve is not assessed. Suppose that for fulfilling a particular function there is a choice between applying 1 kg of resource A or 1 kg of resource B, both of which have an R/P of 20 years. Solely on the basis of these data, resource A and B appear to be equally attractive in terms of depletion. However, closer inspection of the R/P data might show that for resource A $R=10^9$ kg (and $P=5\times10^7$ kg/yr) and for resource B $R=100$ kg (and $P=5$ kg/yr). It is now readily seen that for fulfillment of the specified function resource A is to be preferred, as the reserve of A is affected far less by 1 kg of extraction.

Both these disadvantages can be overcome by taking the reserve into account more than once, for instance by putting the square of the reserve in the denominator of Eqn. (3). It can legitimately be asked whether a higher, e.g cubic, power might not in fact be preferable, as expressed in the following general equation:

$$depletion(\mathrm{kg^{y-z+1} \cdot yr^{-y}}) = \sum_{res} \frac{(deaccumulation_{res}(\mathrm{kg \cdot yr^{-1}}))^y}{(reserve_{res}(\mathrm{kg}))^z} \times extraction_{res}(\mathrm{kg}) \tag{4}$$

It can be proved that the aforementioned dimension requirement leads to $z = y+1$ (ρ is the density of a resource):

$$\begin{aligned}
&\sum_{res} \frac{(deaccumulation_{res}(\mathrm{m^3 \cdot yr^{-1}}))^y}{(reserve_{res}(\mathrm{m^3}))^z} \times extraction_{res}(\mathrm{m^3}) = \\
&= \sum_{res} \frac{(deaccumulation_{res}(\mathrm{kg \cdot yr^{-1}}))^y / (\rho_{res}(\mathrm{kg \cdot m^{-3}}))^y}{(reserve_{res}(\mathrm{kg}))^z / (\rho_{res}(\mathrm{kg \cdot m^{-3}}))^z} \times \frac{extraction_{res}(\mathrm{kg})}{\rho_{res}(\mathrm{kg \cdot m^{-3}})} = \\
&= \sum_{res} \frac{(deaccumulation_{res}(\mathrm{kg \cdot yr^{-1}}))^y}{(reserve_{res}(\mathrm{kg}))^z} \times \frac{extraction_{res}(\mathrm{kg})}{(\rho_{res}(\mathrm{kg \cdot m^{-3}}))^{z-y+1}} = \\
&= \sum_{res} \frac{(deaccumulation_{res}(\mathrm{kg \cdot yr^{-1}}))^y}{(reserve_{res}(\mathrm{kg}))^z} \times extraction_{res}(\mathrm{kg}) \times (\rho_{res}(\mathrm{kg \cdot m^{-3}}))^{y-z+1}
\end{aligned} \tag{5}$$

This equation is equal to Eqn. (4) and is thus independent of ρ if and only if the power $z = y+1$. It has already been argued that production should be included in the equation ($y>0$). Together

with the dimension requirement, this implies that any value of z larger than unity might be appropriate. Curiously, Fava *et al.* [22] give two alternatives for a possible equivalency factor which both meet the requirement, although the requirement itself is not explicitly stated in their report.

This technical discussion of the power in an equation is basically concerned with a very specific problem: assessment of the depletion problem can never be completely verified empirically. For many other types of impacts, equivalency factors can be derived empirically and verified in laboratory or field experiments or by chemical laws. For example, one can verify whether the relation between a concentration of a chemical at site A is linearly, quadratically or inversely proportional to the emission of that chemical at site B, and whether certain impacts on species X occur or not. For resource depletion, another type of reasoning is required. There is no empirically correct method for aggregating extractions of resources to one overall depletion score that can be verified experimentally. In this case, it is necessary to design methods on the basis of logical-theoretical reasoning. It is often possible to test methods by detecting logical contradictions, but one cannot truly validate a non-empirical method.

Any further determination of a value for z (and thus y) on rational grounds seems impossible at present. We can only make a practical suggestion and define $z = 2$ and $y = 1$, thereby confirming Heijungs *et al.* [2] and Fava *et al.* [22]. For a resource expressed in unit u, the effect score then reads:

$$depletion(\mathrm{yr}^{-1}) = \sum_{res} \frac{deaccumulation_{res}(u \cdot \mathrm{yr}^{-1})}{(reserve_{res}(u))^2} \times extraction_{res}(u) \qquad (6)$$

A practical problem associated with this method concerns the sensitivity of the result to the accuracy of the reserves data. The data sources mentioned in the previous section do not give the margins of uncertainty of the ultimate reserves data. As these data become available, the sensitivity of the result to these uncertainties should be further assessed.

5.7 THE PROPOSED EQUIVALENCY FACTORS AND EFFECT SCORES

In order to separately assess differences between biotic and abiotic resources, two separate depletion scores are proposed. Similar to the definition of the GWP [14], depletion potentials can be defined for abiotic resources (ADP=*abiotic depletion potential*) and for biotic resources (BDP=*biotic depletion potential*). Both the ADP and the BDP are defined relative to a reference resource. The concept of ADP and BDP is not needed *per se*, but it has the practical advantage that values thus calculated are much closer to unity and thus easier to comprehend than values calculated by the equivalency factor in Eqn. (6).

In contrast to the GWP, for example, the resource depletion potentials need not be dimensionless, but can optionally be expressed relative to the reference resource in its most suitable unit. As an example, the ADP of natural gas relative to antimony can be expressed in kg antimony per m^3 natural gas or in kg antimony per kg natural gas.

With the following definitions

$$ADP_{res}(u_{ref} \cdot u_{res}^{-1}) = \frac{production_{res}(u_{res} \cdot yr^{-1})}{production_{ref}(u_{ref} \cdot yr^{-1})} \times \left[\frac{reserve_{ref}(u_{ref})}{reserve_{res}(u_{res})} \right]^2 \quad (7)$$

and

$$BDP_{res}(u_{ref} \cdot u_{res}^{-1}) = \frac{deaccumulation_{res}(u_{res} \cdot yr^{-1})}{deaccumulation_{ref}(u_{ref} \cdot yr^{-1})} \times \left[\frac{reserve_{ref}(u_{ref})}{reserve_{res}(u_{res})} \right]^2 \quad (8)$$

where u_{res} denotes the unit in which *res* is expressed and u_{ref} denotes the unit in which *ref* is expressed, aggregation in two scores proceeds by means of

$$equivalent\ abiotic\ use(u_{ref}) = \sum_{res} ADP_{res}(u_{ref} \cdot u_{res}^{-1}) \times extraction_{res}(u_{res}) \quad (9)$$

and

$$equivalent\ biotic\ use(u_{ref}) = \sum_{res} BDP_{res}(u_{ref} \cdot u_{res}^{-1}) \times extraction_{res}(u_{res}) \quad (10)$$

Note that the definition of the ADP contains the production instead of the deaccumulation rate, as abiotic resources are assumed to have zero regeneration.

5.8 EXAMPLE OF APPLICATION WITHIN LCA

ADPs and BDPs can be calculated by applying Eqn. (7) and using data on reserves, production and regeneration rates (see appendix 3 on p. 107 and appendix 4 on p. 108 for preliminary tables of ADP- and BDP-values).

To illustrate how these ADPs and BDPs might be applied in an LCA study, consider a hypothetical case study concerning the packaging of 100 litres of a beverage in PVC- and PE-bottles. This example was used earlier in a paper dealing with the LCA methodology [8]. The resources needed for 100 imaginary PVC-bottles (20 g/bottle) and PE-bottles (50 g/bottle) were computed using the SimaPro 1.0 software [45]. The results of characterizing these extractions using the ADP-concepts are summarized in Table 2.

In this way, two alternative product systems fulfilling the same function can be compared in terms of their relative contributions to resource depletion, on a parallel to a comparison of their relative contributions to global warming and other impact categories. In the hypothetical case study, it is found that with respect to abiotic depletion the PVC-bottle (0.85 kg antimony-equivalents) is environmentally preferable to the PE-bottle (2.62 kg antimony-equivalents).

Table 2: Inventory table of 100 *imaginary* polyethylene bottles (50 g/bottle) and hundred *imaginary* polyvinylchloride bottles (20 g/bottle).

raw material	use PVC bottle (kg)*	use PE bottle (kg)*	ADP (kg/kg)*	ADP × use PVC bottle (kg)	ADP × use PE bottle (kg)
brine (NaCl)	2.03	0			
-chlorine	1.23	0	4.86×10^{-8}	5.99×10^{-8}	0
-sodium	7.99×10^{-1}	0	8.24×10^{-11}	6.59×10^{-11}	0
total coal	5.35×10^{-1}	0	3.63×10^{-3}	1.94×10^{-3}	0
gas	1.35 (m^3)	1.28 (m^3)	3.20×10^{-1}(m^3/kg)	4.34×10^{-1}	4.10×10^{-1}
oil	9.58×10^{-1}	5.08	4.36×10^{-1}	4.17×10^{-1}	2.21
uranium	3.00×10^{-3}	0	2.87×10^{-3}	8.61×10^{-6}	0
TOTAL				0.85	2.62

0 = no data given, which may either mean an actual zero or not measured; this is unclear in most data sources.

* = unless noted otherwise.

5.9 CONCLUSIONS

In this paper we have presented a method, within the framework of LCA, to measure the seriousness of depletion of a given resource in relation to the depletion of another resource.

One conclusion is that it is impossible to define one correct method, since the correctness of the equations involved cannot be verified empirically. On the one hand, this applies to the exact definition of the depletion problem (narrow vs. broad sense); this could, in future, lead to a preference for an entropy-based characterization [27]. On the other hand, this applies to setting up the equations for calculating equivalency factors for depletion, in particular to the power $z = 2$ which has now been "heuristically" derived.

Another conclusion is that with respect to abiotic depletion, data about the most suitable type of reserve, the ultimately extractable reserve, will by definition never be fully available. For the second best type of reserve data, the ultimate reserve, data are sufficiently available. Production data are also still lacking for quite a number of abiotic resources (see notes below table in appendix 4, p. 108). With respect to biotic depletion, much more data are required before a comprehensive list of equivalency factors can be developed.

The method presented in this paper represents one way of characterizing resource extractions within the framework of LCA. There are other possibilities (for an overview, see Finnveden [27]). On the other hand, it is noted that there is also obvious convergence between the methods presented in Heijungs *et al.* [2], Fava *et al.* [22] and the present paper. The subject of resource characterization should thus be submitted to the scientific debate, with the aim of improving the various methods and gathering the missing data.

5.10 ACKNOWLEDGEMENTS

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5.12 APPENDIX 1

Suppose we have two possible definitions of the equivalency factor (E_{res}) for resource *res*, the first on a mass basis:

$$E_{res}(\mathrm{yr}^{-1}) = \frac{deaccumulation_{res}(\mathrm{kg \cdot yr^{-1}})}{reserve_{res}(\mathrm{kg})} \tag{11}$$

and the second on a volume basis:

$$\begin{aligned} E'_{res}(\mathrm{yr}^{-1}) &= \frac{deaccumulation_{res}(\mathrm{m^3 \cdot yr^{-1}})}{reserve_{res}(\mathrm{m^3})} \\ &= E_{res}(\mathrm{yr}^{-1}) \times \frac{\rho_{res}(\mathrm{kg \cdot m^{-3}})}{\rho_{res}(\mathrm{kg \cdot m^{-3}})} \\ &= E_{res}(\mathrm{yr}^{-1}) \end{aligned} \tag{12}$$

Clearly the equivalency factor is insensitive to this change of units.

The effect score (*ES*) on a mass basis is now:

$$ES\ (\mathrm{kg \cdot yr^{-1}}) = \sum_{res} E_{res}\ (\mathrm{yr}^{-1}) \times m_{res}\ (\mathrm{kg}) \tag{13}$$

m_{res} being the amount of resource *res* used in a particular product system.

The effect score (*ES'*) on a volume basis is defined as:

$$\begin{aligned} ES'\ (\mathrm{m^3 \cdot yr^{-1}}) &= \sum_{res} E'_{res}\ (\mathrm{yr}^{-1}) \times V_{res}\ (\mathrm{m^3}) \\ &= \sum_{res} E_{res}(\mathrm{yr}^{-1}) \times \frac{m_{res}(\mathrm{kg})}{\rho_{res}(\mathrm{kg \cdot m^{-3}})} \end{aligned} \tag{14}$$

The requirement that the choice of units should not determine the ultimate appraisal of two product systems (A and B) can be translated into the following condition:

$$\frac{ES_A}{ES_B} = \frac{ES'_A}{ES'_B} \tag{15}$$

ES'_A and ES'_B are connected, respectively, with ES_A and ES_B through a rescaling per resource by a factor ρ_{res}. As the types of resources *res* used will differ for A and B and ρ_{res} therefore differs for the two systems, the condition of Eqn. (15) will in general not be met. It can also be proven that this requirement is still not met when a reference source is introduced, as for the ADP and BDP.

In contrast, it can be proven that for the GWP, for example, this requirement is met. Suppose we have two definitions of the GWP similar to the example above. Then the first is on a mass basis:

$$GWP_i(\mathrm{kg \cdot kg^{-1}}) = \frac{\dfrac{radiative\ forcing_i}{m_i(\mathrm{kg})}}{\dfrac{radiative\ forcing_{ref}}{m_{ref}(\mathrm{kg})}} \tag{16}$$

and the second is on a volume basis:

$$GWP_i'(\mathrm{m^3 \cdot m^{-3}}) = \frac{\dfrac{radiative\ forcing_i}{V_i(\mathrm{m^3})}}{\dfrac{radiative\ forcing_{ref}}{V_{ref}(\mathrm{m^3})}} = GWP_i(\mathrm{kg \cdot kg^{-1}}) \times \frac{\rho_i}{\rho_{ref}} \tag{17}$$

The GWPs of different substances will therefore be different from the original definition.

The global warming effect score (GW) on a mass basis is now:

$$GW(\mathrm{kg}) = \sum_i GWP_i(\mathrm{kg \cdot kg^{-1}}) \times m_i(\mathrm{kg}) \tag{18}$$

m_i being the amount of emission i emitted by a particular product system.

The effect score (GW') on a volume basis is defined as:

$$\begin{aligned} GW'(\mathrm{m^3}) &= \sum_i GWP_i'(\mathrm{m^3 \cdot m^{-3}}) \times V_i(\mathrm{m^3}) \\ &= \sum_i GWP_i(\mathrm{kg \cdot kg^{-1}}) \times \frac{\rho_i(\mathrm{kg \cdot m^{-3}})}{\rho_{ref}(\mathrm{kg \cdot m^{-3}})} \times \frac{m_i(\mathrm{kg})}{\rho_i(\mathrm{kg \cdot m^{-3}})} \\ &= \frac{GW(\mathrm{kg})}{\rho_{ref}(\mathrm{kg \cdot m^{-3}})} \end{aligned} \tag{19}$$

Clearly the condition

$$\frac{GW_A}{GW_B} = \frac{GW_A'}{GW_B'} \tag{20}$$

holds, as GW' can be derived from GW by multiplying the first by a constant (ρ_{ref}).

5.13 APPENDIX 2

For crude oil, coal and natural gas the average carbon content ($C\ content_{fossil\ fuel}$) is known [39]. Proven reserve data of these fossil fuels ($reserve_{proven,fossil\ fuel}$) are known [37]. With these data it can be calculated how much carbon is contained by the total fossil fuel proven reserves:

$$C\ content_{proven,total}(\text{kg}) = \sum_{fossil\ fuel} C\ content_{fossil\ fuel}(\text{kg} \cdot \text{MJ}^{-1}) \times reserve_{proven,fossil\ fuel}(\text{MJ}) \qquad (21)$$

The results are shown in the table:

fossil fuel	$reserve_{proven}$ (MJ)	$C\ content_{fossil\ fuel}$ (kg/MJ)	$C\ content_{proven}$ (kg)
oil	5.17×10^{15}	0.019	9.83×10^{13}
gas	4.25×10^{15}	0.0138	5.86×10^{13}
hard coal	3.00×10^{16}	0.025	7.50×10^{14}
soft coal	7.29×10^{15}	0.0262	1.91×10^{14}
TOTAL			1.10×10^{15}

The total carbon mass in the proven reserves of fossil fuels thus amounts to 1.1×10^{15} kg carbon.

Berner *et al.* [38] estimated the total carbon mass in fossil fuels applying geological theories ($C\ content_{geological}$) to be 1.50×10^{19} kg. This carbon mass (as kerogen) is here assumed to be representing the ultimate reserves of fossil fuel. A rough indication of the fossil fuel ultimate reserves can now be derived by multiplying the proven reserve of resp. oil, gas, soft and hard coal with the ratio between Berner's estimate of the total carbon mass in fossil fuels and the calculated total carbon mass in the proven reserves of these fossil fuels, the ratio being 13.6×10^{3}:

$$reserve_{ultimate,fossil\ fuel}(\text{kg}) = \frac{C\ content_{geological}(\text{kg})}{C\ content_{proven,total}(\text{kg})} \times reserve_{proven,fossil\ fuel}(\text{kg}) \qquad (22)$$

This results in the following ultimate reserves for fossil fuels:

fossil fuel	$reserve_{ultimate}$
crude oil	1.69×10^{18} (kg)
natural gas	1.49×10^{18} (m^3)
soft coal	7.14×10^{18} (kg)
hard coal	1.47×10^{19} (kg)

5.14 APPENDIX 3

BDPs* for a number of species with African elephant as reference species (sources: [40,41,42])

resource	deaccumulation (pieces.yr^{-1})	reserve (pieces)	BDP
african elephant**	1.5×10^{4}	6.0×10^{5}	1.0
black rhino	5.0×10^{2}	3.5×10^{3}	9.8×10^{2}
sperm whale	1.9×10^{2}	1.0×10^{4}	4.6×10^{1}
humpback whale	2.0	4.0×10^{3}	3.0
fin whale	8.9×10^{1}	2.0×10^{3}	5.3×10^{2}
tropical hardwood***	1.83×10^{7}	3.06×10^{9}	4.7×10^{-5}

Notes:

* As stated in the introduction, *normalization* is the final step in the characterization of resource use. For this, the world annual biotic depletion can be calculated by applying Eqn. (9) and (10) and replacing the term $extraction_{res}$ in these equations by the world (annual) production or the world (annual) deaccumulation [cf. 5]. As the normalization aims at relating a product system's effect scores to the world extent of the problem, which in this case is determined by the amount of overexploitation, the world (annual) deaccumulation is to be preferred. This step can easily be made with the data provided in the table above, although it must be stressed that for biotic resources only a very limited set of data is available and that, consequently, the world annual biotic depletion score is highly provisional. Much more data are needed to develop a comprehensive list of BDPs.
The preliminary world annual biotic depletion amounts to 5.47×10^{5} African elephant-equivalents/yr.

** Deaccumulation is estimated based on trends of the past 10-20 years as reported in [41].

*** The BDP of tropical hardwood is relatively low. This is due to the fact that these figures, although the best available, are probably far below actual rates of deforestation [42], and to the fact that depletion only concerns the reserve and deaccumulation of tropical hardwood as a resource and not the destruction of the tropical forest ecosystem. This is not a weakness of this method but, as stated before, it is not a depletion issue: the destruction of the tropical forest ecosystem should be separately assessed, e.g. under the heading of land use, and is actually much more important in relation to tropical deforestation than the resource aspect.

5.15 APPENDIX 4

ADPs* for selected elements and fossil fuels with antimony as reference element[1)]

resource	production (kg/yr)** [28,37]	concentration$_{crust}$ (kg/kg)** [36]	concentration$_{oceans}$ (kg/L)** [36]	concentration$_{atmosphere}$ (kg/kg)** [35]	reserve$_{ultimate}$*** (kg)**	ADP (-)**
actinium (Ac)[3)]	2.90×10^{4}	5.50×10^{-16}	0.00	0.00	1.27×10^{7}	6.33×10^{13}
aluminium (Al)	1.05×10^{11}	8.32×10^{-2}	1.00×10^{-8}	0.00	1.93×10^{21}	1.00×10^{-8}
antimony (Sb)	6.06×10^{7}	2.00×10^{-7}	3.30×10^{-10}	0.00	4.63×10^{15}	1.00
argon (Ar)[3)]	2.90×10^{4}	3.50×10^{-6}	6.00×10^{-7}	1.28×10^{-2}	1.48×10^{17}	4.71×10^{-7}
arsenic (As)	4.50×10^{7}	1.80×10^{-6}	3.00×10^{-9}	0.00	4.17×10^{16}	9.17×10^{-3}
barium (Ba)[3)]	2.90×10^{4}	4.25×10^{-4}	3.00×10^{-8}	0.00	9.84×10^{18}	1.06×10^{-10}
beryllium (Be)	3.79×10^{5}	2.80×10^{-6}	6.00×10^{-13}	0.00	6.48×10^{16}	3.19×10^{-5}
bismuth (Bi)	3.20×10^{6}	1.70×10^{-7}	1.70×10^{-11}	0.00	3.94×10^{15}	7.31×10^{-2}
boron (B)[5)]	7.45×10^{8}	1.00×10^{-5}	4.60×10^{-6}	0.00	2.38×10^{17}	4.67×10^{-3}
bromine (Br)	4.00×10^{8}	2.50×10^{-6}	6.50×10^{-5}	0.00	1.46×10^{17}	6.67×10^{-3}
cadmium (Cd)	2.00×10^{7}	2.00×10^{-7}	1.10×10^{-10}	0.00	4.63×10^{15}	3.30×10^{-1}
calcium (Ca)[5)]	1.85×10^{9}	4.15×10^{-2}	4.00×10^{-4}	0.00	9.61×10^{20}	7.08×10^{-10}
cerium (Ce)[3)]	2.90×10^{4}	6.00×10^{-5}	4.00×10^{-10}	0.00	1.39×10^{18}	5.32×10^{-9}
cesium (Cs)[3)]	2.90×10^{4}	1.00×10^{-6}	5.00×10^{-10}	0.00	2.31×10^{16}	1.91×10^{-5}
chlorine (Cl)[5)]	1.13×10^{8}	1.30×10^{-4}	1.90×10^{-2}	0.00	2.87×10^{19}	4.86×10^{-8}
cobalt (Co)	2.48×10^{7}	2.50×10^{-5}	2.70×10^{-10}	0.00	5.79×10^{17}	2.62×10^{-5}

resource	production (kg/yr)** [28,37]	$concentration_{crust}$ (kg/kg)** [36]	$concentration_{oceans}$ (kg/L)** [36]	$concentration_{atmosphere}$ (kg/kg)** [35]	$reserve_{ultimate}$*** (kg)**	ADP (-)**
copper (Cu)	8.90×10^{9}	5.50×10^{-5}	3.00×10^{-9}	0.00	1.27×10^{18}	1.94×10^{-3}
chromium (Cr)	1.30×10^{10}	1.00×10^{-4}	5.00×10^{-11}	0.00	2.31×10^{18}	8.58×10^{-4}
dysprosium (Dy)[3]	2.90×10^{4}	3.00×10^{-6}	0.00	0.00	6.94×10^{16}	2.13×10^{-6}
erbium (Er)[3]	2.90×10^{4}	2.80×10^{-6}	0.00	0.00	6.48×10^{16}	2.44×10^{-6}
europium (Eu)[3]	2.90×10^{4}	1.20×10^{-6}	0.00	0.00	2.78×10^{16}	1.33×10^{-5}
fluorine (F)[5]	1.75×10^{9}	6.25×10^{-4}	1.30×10^{-6}	0.00	1.45×10^{19}	2.96×10^{-6}
gadolinium (Gd)[3]	2.90×10^{4}	5.40×10^{-6}	0.00	0.00	1.25×10^{17}	6.57×10^{-7}
gallium (Ga)	3.50×10^{4}	1.50×10^{-5}	3.00×10^{-11}	0.00	3.47×10^{17}	1.03×10^{-7}
germanium (Ge)	6.50×10^{4}	5.40×10^{-6}	7.00×10^{-11}	0.00	1.25×10^{17}	1.47×10^{-6}
gold (Au)	2.17×10^{6}	4.00×10^{-9}	1.10×10^{-11}	0.00	9.26×10^{13}	8.95×10^{1}
hafnium (Hf)[3]	2.90×10^{4}	4.70×10^{-6}	8.00×10^{-12}	0.00	1.09×10^{17}	8.67×10^{-7}
helium (He)	1.49×10^{7}	8.00×10^{-9}	6.90×10^{-12}	7.00×10^{-7}	1.89×10^{14}	1.48×10^{2}
holmium (Ho)[3]	2.90×10^{4}	1.20×10^{-6}	0.00	0.00	2.78×10^{16}	1.33×10^{-5}
indium (In)	1.40×10^{5}	1.00×10^{-7}	2.00×10^{-8}	0.00	2.34×10^{15}	9.03×10^{-3}
iodine (I)	1.64×10^{7}	5.00×10^{-7}	6.00×10^{-8}	0.00	1.17×10^{16}	4.27×10^{-2}
iridium (Ir)[4]	4.90×10^{4}	1.00×10^{-9}	0.00	0.00	2.31×10^{13}	3.23×10^{1}
iron (Fe)	4.05×10^{11}	5.63×10^{-2}	1.00×10^{-8}	0.00	1.30×10^{21}	8.43×10^{-8}
kalium (K)	2.08×10^{10}	2.09×10^{-2}	3.80×10^{-4}	0.00	4.84×10^{20}	3.13×10^{-8}
krypton (Kr)[3]	2.90×10^{4}	1.00×10^{-10}	2.50×10^{-9}	3.20×10^{-6}	2.21×10^{13}	2.09×10^{1}
lanthanum (La)[3]	2.90×10^{4}	3.00×10^{-5}	1.20×10^{-11}	0.00	6.94×10^{17}	2.13×10^{-8}
lithium (Li)	5.60×10^{6}	2.00×10^{-5}	1.80×10^{-7}	0.00	4.63×10^{17}	9.23×10^{-6}

resource	production (kg/yr)** [28,37]	concentration$_{crust}$ (kg/kg)** [36]	concentration$_{oceans}$ (kg/L)** [36]	concentration$_{atmosphere}$ (kg/kg)** [35]	reserve$_{ultimate}$*** (kg)**	ADP (-)**
lead (Pb)	3.20×10^{9}	1.25×10^{-5}	3.00×10^{-11}	0.00	2.89×10^{17}	1.35×10^{-2}
lutetium (Lu)[3]	2.90×10^{4}	5.00×10^{-7}	0.00	0.00	1.16×10^{16}	7.66×10^{-5}
magnesium (Mg)	3.09×10^{9}	2.33×10^{-2}	1.35×10^{-3}	0.00	5.41×10^{20}	3.73×10^{-9}
manganese (Mn)	1.88×10^{10}	9.50×10^{-4}	2.00×10^{-9}	0.00	2.20×10^{19}	1.38×10^{-5}
mercury (Hg)	4.80×10^{6}	8.00×10^{-8}	3.00×10^{-11}	0.00	1.85×10^{15}	4.95×10^{-1}
molybdenum (Mo)	1.08×10^{8}	1.50×10^{-6}	1.00×10^{-8}	0.00	3.47×10^{16}	3.17×10^{-2}
neodymium (Nd)	2.30×10^{-5}	2.80×10^{-5}	0.00	0.00	6.48×10^{17}	1.94×10^{-17}
neon (Ne)[3]	2.90×10^{4}	5.00×10^{-9}	1.40×10^{-10}	1.20×10^{-5}	1.78×10^{14}	3.25×10^{-1}
nickel (Ni)	9.16×10^{8}	7.50×10^{-5}	5.40×10^{-9}	0.00	1.74×10^{18}	1.08×10^{-4}
niobium (Nb)	1.40×10^{7}	2.00×10^{-5}	1.00×10^{-11}	0.00	4.63×10^{17}	2.31×10^{-5}
osmium (Os)[4]	4.90×10^{4}	1.50×10^{-9}	0.00	0.00	3.47×10^{13}	1.44×10^{1}
palladium (Pd)[4]	4.90×10^{4}	1.00×10^{-8}	0.00	0.00	2.31×10^{14}	3.23×10^{-1}
phosphorus (P)	1.41×10^{11}	1.05×10^{-3}	7.00×10^{-8}	0.00	2.43×10^{19}	8.44×10^{-5}
platinum (Pt)[4]	4.90×10^{4}	5.00×10^{-9}	0.00	0.00	1.16×10^{14}	1.29
polonium (Po)[3]	2.90×10^{4}	2.00×10^{-16}	0.00	0.00	4.63×10^{6}	4.79×10^{14}
praseodymium (Pr)[3]	2.90×10^{4}	8.20×10^{-6}	0.00	0.00	1.90×10^{17}	2.85×10^{-7}
protactinium (Pa)[3]	2.90×10^{4}	1.40×10^{-12}	2.00×10^{-15}	0.00	3.24×10^{10}	9.77×10^{6}
radium (Ra)[3]	2.90×10^{4}	9.00×10^{-13}	6.00×10^{-17}	0.00	2.08×10^{10}	2.36×10^{7}
radon (Rn)[3]	2.90×10^{4}	4.00×10^{-19}	6.00×10^{-22}	0.00	9.26×10^{3}	1.20×10^{20}
rhenium (Re)	2.90×10^{4}	5.00×10^{-9}	0.00	0.00	1.16×10^{14}	7.66×10^{-1}
rhodium (Rh)[4]	4.90×10^{4}	1.00×10^{-9}	0.00	0.00	2.31×10^{13}	3.23×10^{1}

resource	production (kg/yr)** [28,37]	concentration$_{crust}$ (kg/kg)** [36]	concentration$_{oceans}$ (kg/L)** [36]	concentration$_{atmosphere}$ (kg/kg)** [35]	reserve$_{ultimate}$*** (kg)**	ADP (-)**
rubidium (Rb)[3]	2.90×10^{4}	9.00×10^{-5}	1.20×10^{-7}	0.00	2.08×10^{18}	2.36×10^{-9}
ruthenium (Ru)[4]	4.90×10^{4}	1.00×10^{-9}	0.00	0.00	2.31×10^{13}	3.23×10^{1}
samarium (Sm)[3]	2.90×10^{4}	6.00×10^{-6}	0.00	0.00	1.39×10^{17}	5.32×10^{-7}
scandium (Sc)[3]	2.90×10^{4}	2.20×10^{-5}	0.00	0.00	5.09×10^{17}	3.96×10^{-8}
selenium (Se)	1.80×10^{6}	5.00×10^{-8}	9.00×10^{-11}	0.00	1.16×10^{15}	4.75×10^{-1}
silicium (Si)	3.60×10^{9}	2.82×10^{-1}	3.00×10^{-6}	0.00	6.53×10^{21}	2.99×10^{-11}
silver (Ag)	1.37×10^{7}	7.00×10^{-8}	3.00×10^{-10}	0.00	1.62×10^{15}	1.84
sodium (Na)[5]	7.32×10^{7}	2.36×10^{-2}	1.05×10^{-2}	0.00	5.60×10^{20}	8.24×10^{-11}
strontium (Sr)	2.38×10^{8}	3.75×10^{-4}	8.10×10^{-6}	0.00	8.69×10^{18}	1.12×10^{-6}
sulfur (S)	5.27×10^{10}	2.60×10^{-4}	8.85×10^{-4}	2.47×10^{-10}	7.21×10^{18}	3.58×10^{-4}
tantalum (Ta)	4.10×10^{5}	2.00×10^{-6}	2.50×10^{-12}	0.00	4.63×10^{16}	6.77×10^{-5}
tellurium (Te)	8.00×10^{4}	1.00×10^{-9}	0.00	0.00	2.31×10^{13}	5.28×10^{1}
terbium (Tb)[3]	2.90×10^{4}	9.00×10^{-7}	0.00	0.00	2.08×10^{16}	2.36×10^{-5}
thallium (Tl)	1.55×10^{4}	4.50×10^{-7}	1.00×10^{-11}	0.00	1.04×10^{16}	5.05×10^{-5}
thorium (Th)[3]	2.90×10^{4}	9.60×10^{-6}	5.00×10^{-11}	0.00	2.22×10^{17}	2.08×10^{-7}
thulium (Tm)[3]	2.90×10^{4}	4.80×10^{-7}	0.00	0.00	1.11×10^{16}	8.31×10^{-5}
tin (Sn)	2.00×10^{8}	2.00×10^{-6}	3.00×10^{-9}	0.00	4.63×10^{16}	3.30×10^{-2}
titanium (Ti)[5]	2.16×10^{9}	5.70×10^{-3}	1.00×10^{-9}	0.00	1.32×10^{20}	4.40×10^{-8}
tungsten (W)	3.98×10^{7}	1.50×10^{-6}	1.00×10^{-10}	0.00	3.47×10^{16}	1.17×10^{-2}
uranium (U)[5]	3.17×10^{7}	2.70×10^{-6}	3.00×10^{-9}	0.00	6.25×10^{16}	2.87×10^{-3}
vanadium (V)	3.21×10^{7}	1.35×10^{-4}	2.00×10^{-9}	0.00	3.13×10^{18}	1.16×10^{-6}

resource	production (kg/yr)** [28,37]	$concentration_{crust}$ (kg/kg)** [36]	$concentration_{oceans}$ (kg/L)** [36]	$concentration_{atmosphere}$ (kg/kg)** [35]	$reserve_{ultimate}$*** (kg)**	ADP (-)**
xenon (Xe)[3]	2.90×10^{4}	3.00×10^{-11}	5.20×10^{-11}	0.00	7.65×10^{11}	1.75×10^{4}
ytterbium (Yb)[3]	2.90×10^{4}	3.00×10^{-6}	0.00	0.00	6.94×10^{16}	2.13×10^{-6}
yttrium (Y)[5]	5.51×10^{5}	3.30×10^{-5}	3.00×10^{-10}	0.00	7.64×10^{17}	3.34×10^{-7}
zinc (Zn)	7.37×10^{9}	7.00×10^{-5}	1.00×10^{-8}	0.00	1.62×10^{18}	9.92×10^{-4}
zirconium (Zr)	7.65×10^{8}	1.65×10^{-4}	2.20×10^{-11}	0.00	3.82×10^{18}	1.86×10^{-5}
crude oil[6]	3.51×10^{12}	nap	nap	nap	1.69×10^{18}	4.36×10^{-1}
natural gas[6,7]	2.02×10^{12}	nap	nap	nap	1.49×10^{18}	3.20×10^{-1}
soft coal (<23.8 MJ/kg)[6]	1.23×10^{12}	nap	nap	nap	7.14×10^{18}	8.51×10^{-3}
hard coal (≥23.8 MJ/kg)[6]	3.66×10^{12}	nap	nap	nap	1.47×10^{19}	6.00×10^{-3}

Sources: [28,36,35,37].

Notes:

* As stated in the introduction *normalization* is the final step in the characterization of resource use. For this, the world annual abiotic depletion can be calculated by applying Eqn. (9) and (10) and replacing the term $extraction_{res}$ in these equations by the world (annual) production or the world (annual) deaccumulation [cf. 5]. As the normalization aims at relating a product system's effect scores to the world extent of the problem, which in this case is determined by the amount of overexploitation, the world (annual) deaccumulation is to be preferred. This step can easily be made with the data provided in the table above. The world annual abiotic depletion amounts to 2.20×10^{12} kg antimony-equivalents/yr

** Unless specified otherwise.

*** The $reserve_{ultimate}$ is calculated as follows:

$$reserve_{ultimate} = (mass_{crust} \times concentration_{crust}) + (volume_{oceans} \times concentration_{oceans}) + (mass_{atmosphere} \times concentration_{atmosphere})$$

$mass_{crust} = 2.31 \times 10^{22}$ kg assuming the average crust thickness to be 17 km [33], the average crust density to be 2500 kg.m^{-3} [36] and the crust surface to be 5.14×10^{14} m^2 [35];
$volume_{oceans} = 1.35 \times 10^{21}$ l [35];
$mass_{atmosphere} = 5.14 \times 10^{18}$ kg [35].

1) Regeneration is assumed to be zero for all abiotic resource types.
2) nap means not applicable.
3) Because specific production data were lacking, it is assumed that the production is equal to the production of rhenium. As there is no solid basis for this asssumption, the reliability of the resulting ADP is low. This assumption can lead to odd results as is the case with radon. By assuming that the production of radon is equal to the production of rhenium, the reserve has become smaller than the (annual) production which is very unlikely.
4) The production value given in [28] actually indicates the sum of the production of platinum, palladium, rhodium, ruthenium, iridium and osmium. Separate production data are not available. The sum value for the production is here divided by six and assumed to be equal for each of the resources mentioned.
5) Production data are often given in tons of complex mineral resources such as for B_2O_3, CaF_2, NaCl, TiO_2, U_3O_8 and Y_2O_3. These complex minerals have to be transposed into their constituting elements based on their relative molecular weights.
6) Reserve data are given in PJ and converted to kg for coal and oil and to m^3 for gas by multiplication with their energy contents which is for oil equal to 41.87 MJ/kg, for hard coal 27.91 MJ/kg, for soft coal 13.96 MJ/kg for gas 38.84 MJ/m^3 [37].
7) Production in m^3.yr^{-1}, reserve in m^3, and ADP in kg.m^{-3}.

Chapter 6

Methodological case study on margarines

6.1 ABSTRACT

A case study has been performed to assess the extent to which the results of an LCA are influenced by choices in methodology and the data used in the various methodological steps and also to illustrate the LCA methodology proposed in the previous chapters. Four different margarines used for frying, roasting and spreading on bread are assessed and compared. A final appraisal of the margarines was not aimed at. In order to determine the influence of choices in methodology and data, sensitivity analyses were performed with respect to different allocation methods for coproduction, different estimates of pesticide emissions, different methods for characterizing extractions of abiotic resources, different data sets for Global Warming Potentials and Photochemical Ozone Creation Potentials, different valuation methods and different impact assessment methods.

Results were calculated using default options with respect to the aforementioned choices of methodology and data. During calculation of the results there was found to be a major error in some of the process data: the emissions of pesticides were shown to be erroneously high. The magnitude of this error was estimated to be about a factor 1000. Sensitivity analyses were performed after correction of this error.

The results appeared to be particularly sensitive to the choice of the impact assessment method. Three impact assessment approaches have been compared: the multi-step "Environmental Themes" approach, the single-step EPS approach and the single-step Ecopoints approach. Use of these three methods gave completely different results with respect to the environmental preference of the four margarines studied. Focusing of the "Environmental Themes" impact assessment methodology, the next most sensitive choice appeared to be the equation used for calculating one of the weighting factors in the valuation. Finally, the results of the study appeared to be quite insensitive to the choice of methods used for allocating coproduction and for characterizing extractions of abiotic resources and for the range of values for GWPs and POCPs.

6.2 INTRODUCTION

In the previous chapters methodological proposals have been developed and discussed for almost all components and steps of environmental LCA of products. Although a number of methodological issues require further study, at the same time it has become clear that, at least to some extent, there will always remain choices within the methodology and shortcomings in the availability and/or reliability of data. These choices and related methods and data can substantially influence the final results of a case study. Therefore, at the end of this thesis it seems appropriate to perform a methodological case study with two specific goals: 1) to assess to what extent the results of a case study can be influenced by choices of methodology and data used in the various methodological steps; and 2) to illustrate the potential applicability of the LCA-methodology proposed in previous chapters.

The subjects of this case study are four margarines (A, B, C and D) produced by the company Van den Bergh Nederland. The database on these margarines stems from April 1992 and was set up in a pilot study on the same four margarines by Vis *et al.* [1]. Data on the production

of energy, electricity and packaging materials and data on the incineration of wastes were taken from Habersatter and Widmer [2] with additional data from Lundholm and Sundström [3]. Data on the production of P_2O_5 were taken from Hoogenkamp [4] and data on landfill of wastes were taken from Sundqvist *et al.* [5]. All data stem from about 1990 except for the additional data from Lundholm and Sundström, which stem from the beginning of the eighties.

Additional data gathering and extensive data quality assessment were not possible within the framework of this thesis. Although the results of this case study may give some indication of the environmental effects of the margarines considered, they cannot be considered decisive.

The study is subdivided into six major sections: goal definition and scoping, inventory analysis, impact assessment, results, sensitivity analyses and conclusions. The first three sections are named and organized in line with the components and steps of the methodological framework defined in Chapter 2 of this thesis [cf. 6], with some adaptations based on the framework as defined by SETAC [7]*. In this case study, no attention has been given to the improvement assessment, as there are as yet no clear choices as to the methodology and data involved in this component. Within a given component, each step is discussed according to the same structure: first a short introduction is given, then options are discussed. If possible, options for methodology and data are subjected to sensitivity analyses. However, where the data or time required for a full sensitivity analysis are lacking, examples have been derived from other studies, if available, in order to provide an indication of the sensitivity of the results. In the results section, results are presented at the level of the ecoprofile, the normalized ecoprofile and the environmental index for a default selection of options. In the section on sensitivity analyses, the default choices as defined in the results have been changed and the influence of these alternative choices calculated at the level of the environmental index. Finally, conclusions are drawn with respect to the results of this particular case study as well as regarding some general aspects of current methodology and data.

6.3 GOAL DEFINITION AND SCOPING

6.3.1 Goal and application

As stated above, the goal of this study is to illustrate the LCA-theory as described in the previous

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* As already noted in the previous chapters, the methodological framework and terminology proposed in Chapter 2 of this thesis differs in some respects from those proposed in the SETAC "Code of Practice" [7]. For reasons of international standardization, the "Code of Practice" framework is followed in this chapter, distinguishing Goal Definition and Scoping, Inventory Analysis, Impact Assessment and Improvement Assessment. However, as further subdivision of these methodological components into steps in the "Code of Practice" was explicitly made for the Impact Assessment only, steps for the other components are largely taken from this thesis [cf. Chapters 2 and 3]. Application of the allocation rules is not distinguished as a separate step of the inventory analysis but is treated as part of the process tree, and the reliability and validity assessments are not treated as steps of the valuation but are the leitmotiv of a separate section dealing with sensitivity analyses. This results in the following framework:

components	steps
Goal Definition and Scoping	Goal and application; Scope and depth of the study; Product systems studied; Functional unit
Inventory Analysis	Process tree; Process data; Inventory table
Impact Asssessment	Classification; Characterization; Valuation
Improvement Assessment	

chapters of this thesis by a case study on margarines and to assess the extent to which the results of a case study can be influenced by choices in methodology and the data used in the various methodological steps. The study was initiated by the author with the support of data on the margarines from Van den Bergh Nederland. The target groups of this study are LCA-methodology developers and LCA-practitioners. As indicated in the introduction, it is not the intention of this case study to make a decisive appraisal of the products involved.

6.3.2 Product systems studied

The product systems studied are four margarines produced for the Dutch market in 1992 by Van den Bergh Nederland. Margarines are an emulsion of water in oil. Both the water and the oil phases contain a number of ingredients such as aromas, lecithin, whey powder, etc. The margarine is packed in tubs or wrappers. It is assumed that the margarines are functionally equivalent [1] and can be used for frying, roasting and spreading on bread. Three of the margarines studied are sold in 250 gram units and one in 500 gram units.

6.3.3 Scope and depth of the study

As it is explicitly not the intention of this study to make a final environmental appraisal of the margarines involved, the study is not an in-depth one with respect to the constituent processes. The process tree of the margarines is not complete and is cut off at several stages. It is not known whether these cut-offs lead to significant errors.

In the original study by Vis *et al.* [1] only the fat component of the margarines was studied, with other ingredients being left out of consideration. This study is also restricted to the fat component.

6.3.4 Functional unit

In the previous study on the same margarines [1] the functional unit was chosen to be 1000 kg of prepared and packed product (net weight). A theoretically better functional unit seems to be the use of 1000 kg of margarine by a consumer, as this gives a better indication of the final function of the margarine. If the use of margarine by an average consumer is also defined for a certain time period and a certain area, e.g. one year and The Netherlands, there are some advantages of presentation which will be adressed in the section on the normalization step. Thus, the functional unit here is "the consumption of margarine by the average consumer in The Netherlands per year", which is assumed to equal 10 kg of margarine per year per person [cf. 8].

This functional unit includes both the life cycle of the margarine and the life cycle of the packaging. Until now, food packaging studies have not usually included both the life cycle of the food product and the life cycle of the packaging, but have focused mainly on the latter. However, the packaging and the product it contains are often related and, as Kooijman has recently argued [9], in some cases separation of these two life-cycles may lead to substantial errors. For example, it may be the case that differences in the adhesive properties of the packaging alternatives give rise to different amounts of food spillage. Moreover, not all sizes of packaging will be appropriate for all situations. For example, it would not be sensible for a single person to buy coffee milk in a large-sized packaging. He or she would probably have to throw away a large part

of the milk as it might go sour before consumption. Nevertheless, whether and how these facts should be taken into account depends on the goal of the assessment. For instance, it depends on the consumer or consumer group (e.g. families or canteen visitors) whether a certain amount of product loss will result in an increased consumption and whether a certain packaging size may lead to premature product deterioration. Thus, there may be different environmental truths for a given product, depending on the functions it can have. It is conceivable that LCA indicates that a household should give preference to small margarine tubs, and canteens to large-sized margarine wrappers. The result appears to be user-dependent and thus context-sensitive [10].

With respect to the present case study, food wastage due to the size of packagings studied seems irrelevant, as margarines do not spoil that fast. Food spillage due to the type of packaging material used has not been analyzed. If data about this relation becomes available, the number of packagings to be compared should be adapted accordingly. For example, if in packaging A 5% more margarine is left behind than in packaging B, one needs to produce 10.5 kg of margarine A but only 10 kg of margarine B according to the functional unit defined above. Kooijman [9] found that these kinds of product losses may amount to 20% for viscous products such as yoghurt.

6.4 INVENTORY ANALYSIS

6.4.1 **Process tree**

6.4.1.1 *Boundary between product system and environment system*

Figure 1 summarizes the process tree used for the four margarines considered in this study. It is intended merely to provide an impression of the general process tree of the margarines studied. The detailed process tree of a single margarine may include well over 100 individual processes and is specific to each margarine; it is beyond the scope of this chapter to present such a detailed process tree.

With respect to the boundaries between the product system and the environment system, the theory described in Chapter 2 and in Heijungs *et al.* [6] is followed. This means, for example, that agriculture, forestry and landfill of waste are considered to be economic processes, with the associated emissions of pesticides and so on. Besides some serious data problems related to these processes (see below), there are two special cases of boundary definition between product systems and environmental systems with respect to agricultural products such as margarines: 1) short and long carbon cycles, and 2) deliberately applied chemicals such as nutrients and pesticides. The case of deliberately applied chemicals is treated below in the section on process data, where a sensitivity analysis is described for pesticides, focusing on the definition of the boundary between agricultural plots and the environment and the related quantification of pesticide emissions.

In this section, the focus is on the case of short and long carbon cycles. Short carbon cycles are taken to mean, for example, the reforestry process, whereby an amount of carbon in the form of CO_2 is fixed during tree growth and is ultimately released in the form CO_2 or CH_4 when the wood is landfilled or incinerated or decays naturally. The most appropriate way to treat short carbon cycles is to view them as cycles and thus subtract the fixation of CO_2 during tree growth from the CO_2 emitted during waste treatment of discarded wood and to quantify the CH_4

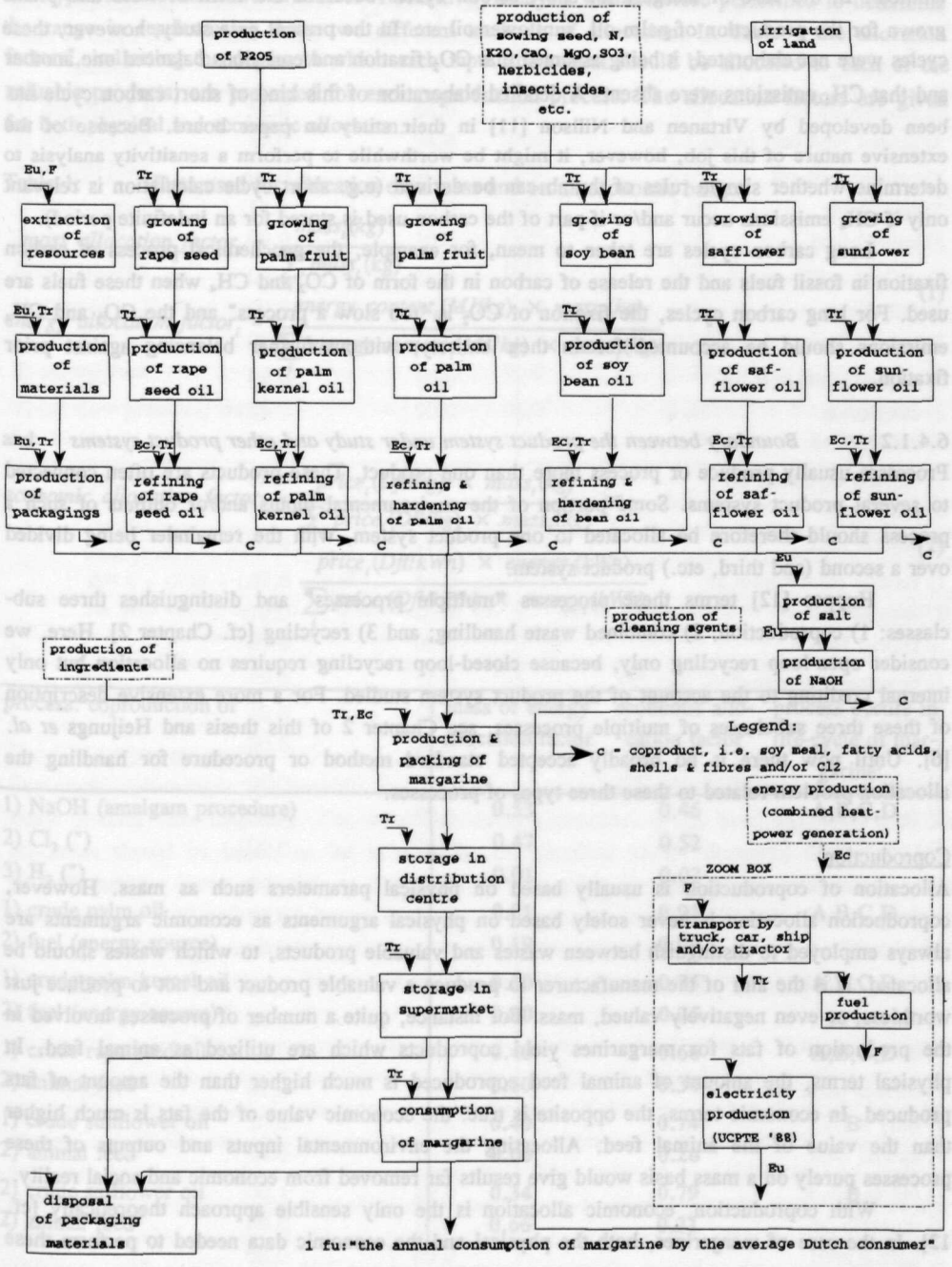

Figure 1: General process tree of the four margarines

emitted*. In the case of margarines, short carbon cycles occur in the form of trees and plants grown for the production of palm oil, sunflower oil etc. In the present case study, however, these cycles were not elaborated, it being assumed that CO_2 fixation and emissions balanced one another and that CH_4 emissions were absent. A detailed elaboration of this kind of short carbon cycle has been developed by Virtanen and Nillson [11] in their study on paper board. Because of the extensive nature of this job, however, it might be worthwhile to perform a sensitivity analysis to determine whether simple rules of thumb can be derived (e.g. short cycle calculation is relevant only if CH_4 emissions occur and/or if part of the carbon used is stored for an indefinite period).

Long carbon cycles are taken to mean, for example, the geochemical process of carbon fixation in fossil fuels and the release of carbon in the form of CO_2 and CH_4 when these fuels are used. For long carbon cycles, the fixation of CO_2 is "too slow a process" and the CO_2 and CH_4 emissions should be accounted for in their entirety, without further balancing against prior fixation.

6.4.1.2 *Boundary between the product system under study and other product systems*

Processes usually produce or process more than one product. These products are often connected to several product systems. Some portion of the environmental inputs and/or outputs of such a process should therefore be allocated to one product system, with the remainder being divided over a second (and third, etc.) product system.

Huppes [12] terms these processes "multiple processes" and distinguishes three subclasses: 1) coproduction; 2) combined waste handling; and 3) recycling [cf. Chapter 2]. Here, we consider open-loop recycling only, because closed-loop recycling requires no allocation but only internal crediting to the account of the product system studied. For a more extensive description of these three subclasses of multiple processes, see Chapter 2 of this thesis and Heijungs *et al.* [6]. Until now there is no broadly accepted standard method or procedure for handling the allocation problem related to these three types of processes.

Coproduction

Allocation of coproduction is usually based on physical parameters such as mass. However, coproduction allocation is never solely based on physical arguments as economic arguments are always employed to distinguish between wastes and valuable products, to which wastes should be allocated. It is the aim of the manufacturer to produce a valuable product and not to produce just worthless, or even negatively valued, mass. For instance, quite a number of processes involved in the production of fats for margarines yield coproducts which are utilized as animal feed. In physical terms, the amount of animal feed coproduced is much higher than the amount of fats produced. In economic terms, the opposite is true: the economic value of the fats is much higher than the value of the animal feed. Allocating the environmental inputs and outputs of these processes purely on a mass basis would give results far removed from economic and social reality.

With coproduction, economic allocation is the only sensible approach theoretically [cf. 12]. In the case of margarines, both the physical and the economic data needed to perform these

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* This balancing of carbon as an element in compounds such as CO_2 and CH_4 is necessary because CO_2 and CH_4 have different equivalency factors with respect to global warming ($GWP_{CO2,100} \div GWP_{CH4,100} = 1 \div 11$).

two types of allocation are available. A sensitivity analysis was therefore performed to determine the range in results using these two different methods of allocation. In Table 1 the allocation factors*, indicating the degree to which each input and output should be allocated to each of the valuable products, are presented for each coproduction process. The allocation factors are given for both physical and economic allocation.

Table 1: Data used for allocation on a mass and on an economic basis:

$$\textit{mass allocation factor}_i = \frac{mass_i(kg)}{\sum_i mass_i(\text{kg})}$$

$$\textit{energy allocation factor}_i = \frac{\textit{energy content}_i(MJ/kg) \times mass_i(kg)}{\sum_i \textit{energy content}_i(MJ/kg) \times mass_i(kg)} \quad (1)$$

and

$$\textit{economic allocation factor}_i = \frac{price_i(Dfl/kg) \times mass_i(kg)}{\sum_i price_i(Dfl/kg) \times mass_i(kg)} \text{ or}$$

$$\frac{price_i(Dfl/kWh) \times energy_i(kWh)}{\sum_i price_i(Dfl/kWh) \times energy_i(kWh)} \quad (2)$$

process: coproduction of	mass or energy allocation factor	economic allocation factor	process occurs in life-cycle of margarine:
1) NaOH (amalgam procedure)	0.53	0.46	A,B,C,D
2) Cl_2 (")	0.47	0.52	
3) H_2 (")	0.01	0.02	
1) crude palm oil	**0.81**	**0.91**	A,B,C,D
2) fuel (energy source)	**0.19**	**0.09**	
1) crude palm kernel oil	**0.50**	**0.75**	A,B,C,D
2) fuel (energy source)	**0.50**	**0.25**	
1) crude rape seed oil	**0.40**	**0.66**	A,B,C,D
2) animal feed	**0.60**	**0.34**	
1) crude sunflower oil	**0.40**	**0.74**	B
2) animal feed	**0.60**	**0.26**	
2) crude safflower oil	**0.34**	**0.79**	B
2) animal feed	**0.66**	**0.21**	

* For reasons of consistency, their sum should be 1.

process: coproduction of	mass or energy allocation factor	economic allocation factor	process occurs in life-cycle of margarine:
1) crude bean oil	**0.20**	**0.32**	B,C
2) animal feed (soy meal)	**0.80**	**0.68**	
1) refined bean oil	0.96	0.99	C,D
2) fatty acids	0.03	0.01	
3) animal feed	0.01	0.00	
1) refined safflower oil	0.96	1.00	B,D
2) fatty acids	0.03	0.00	
3) animal feed	0.01	0.00	
1) refined sunflower oil	0.97	0.99	B
2) fatty acids	0.02	0.01	
3) animal feed	0.01	0.00	
1) refined rape seed oil	0.96	0.99	A,B,C,D
2) fatty acids	0.03	0.01	
3) animal feed	0.01	0.00	
1) refined palm kernel oil	0.98	0.99	A
2) fatty acids	0.02	0.01	
3) animal feed	0.00	0.00	
1) processed rape seed oil (mp 36)	0.97	0.99	A,D
2) fatty acids	0.02	0.01	
3) animal feed	0.01	0.00	
1) processed rape seed oil (mp 28)	0.97	0.99	A
2) fatty acids	0.02	0.01	
3) animal feed	0.01	0.00	
1) hardened palm oil (mp 44)	0.92	0.98	A,B,C
2) fatty acids	0.07	0.02	
3) animal feed	0.01	0.00	
1) inesterified palm oil-palm kernel oil mix	0.92	0.98	A,C,D
2) fatty acids	0.07	0.02	
3) animal feed	0.01	0.00	
1) refined palm oil	0.94	0.98	A,B,C
2) fatty acids	0.05	0.02	
3) animal feed	0.01	0.00	
1) margarine A	0.98	1.00	A
2) reusable fat	0.02	0.00	
3) animal feed	0.00	0.00	

process: coproduction of	mass or energy allocation factor	economic allocation factor	process occurs in life-cycle of margarine:
1) margarine B	0.94	0.99	B
2) reusable fat	0.06	0.01	
3) animal feed	0.00	0.00	
1) margarine C	0.95	1.00	C
2) rework fat	0.05	0.00	
3) animal feed	0.00	0.00	
1) margarine D	0.98	1.00	D
2) reusable fat	0.02	0.00	
3) animal feed	0.00	0.00	
1) LP (low pressure) steam	**0.63***	**0.39****	A,B,C,D
2) electricity	**0.37***	**0.61****	A,B,C,D

* In the cogeneration of steam and electricity, 1000 kg of LP steam with an energy content of 2635 MJ [13] and 431 kWh representing 431 kWh × 3.6 MJ/kWh = 1551.6 MJ is produced. So the relative energy content of the steam produced is 2635 ÷ (2635 + 1551.6) = 0.63 and the relative energy content of the electricity produced is 1551.6 ÷ (2635 + 1551.6) = 0.37.

** The price of 1000 kg LP steam is assumed to be Dfl. 21.91 and the price of 431 kWh electricity is assumed to be Dfl. 34.48. The ratio of the economic allocation factors for LP steam:electricity is 0.39:0.61.

*** Bold typed figures indicate that mass/energy and economic allocation factors differ more than 10%.

Combined waste processing

Communal incinerators and landfills are examples of processes which process a wide range of waste streams simultaneously. For combined waste processing it has been suggested that the allocation should be based, as far as possible, on (physical and/or chemical) causal relations between process inputs and outputs [5,6,12,15]. The suggestion may be correct for both waste incineration as well as landfill, but to implement this approach is another matter. Causal relations in a communal waste incinerator are not that clear, for example, as is shown by the ongoing discussions about the relation between PVC in household waste and dioxin emitted from incinerators.

Allocation based on causal relations can be made as complicated as desired. A simple starting point might be that all emissions of chemical compounds must have a chemical origin. In other words, an emission of carbon dioxide can only occur if there is a carbon source in the waste stream, and dioxin can only be emitted if there are sources of both carbon and chlorine. All emissions are then considered to be product-derived. This approach was followed in a study on foam packaging [14]. A more sophisticated approach is to distinguish, for example, two categories of emissions [cf. 5,15]: 1) product-derived and 2) process-derived. Product-derived emissions, such as CO_2, SO_2, HCl and heavy metals, are mainly related to the composition of the waste products incinerated (e.g. the presence of carbon, sulphur, chlorine and heavy metals in the product) and should be allocated on this basis. Process-derived emissions such as PAH, dioxin and

CO are primarily related to the process conditions and not to the composition of the waste products incinerated. Although the sophisticated approach is the better one in theory, it is very hard, if at all possible, to allocate the process-derived emissions. Only a few initial attempts to do so are known, but these have not yet resulted in practical data sets for process-derived emissions [5,15]. Data sets are now limited to product-derived bulk emissions such as CO_2, SO_2 and HCl [2]. According to a first estimate of Guinée *et al.* [14], a process-derived emission such as PAH allocated to the incineration of a bulk material such as polyethylene [14] might amount to almost 0.1 mg/kg incinerated, based upon average Dutch technology.

For a landfill the same discussion is more or less valid, with one addition. In determining the emission data of landfills one has to specify the time-frame to be considered, as landfill emissions may persist for a very long time (>100 years) [5,6]. The most ideal situation would be if all potential emissions, including all transformation products of the initial chemicals emitted, could be predicted as a function of time [5,6]. However, the necessary knowledge is simply not available. Rougher estimates must therefore be made, based on the limited knowledge that is available.

Sundqvist *et al.* [5] recently performed initial estimates of a number of emissions occurring from a landfill over a 100-year period. As they themselves state, this initial attempt needs further improvement with respect to both the allocation principles applied and the time-frame considered. They proposed two time-frames: the surveyable time-perspective (100 years) and the hypothetical, infinite time-perspective (infinite number of years). Suppose that the annual average leaching of some metal amounts to 1% of the amount landfilled, then the estimated emission over the surveyable time-perspective amounts to 63%, while that over the infinite time-perspective would be 100% of the amount landfilled. If, however, the annual average leaching of that metal amounts to only 0.1% of the amount landfilled, then the estimated emissions would be 9.5% and 100%, respectively. From this example it is clear that the estimate of annual leaching in relation to the time-frame considered may considerably influence estimates of landfill emissions and, as a consequence, also the results of a particular case study*.

Open-loop recycling

Open-loop recycling occurs when waste produced by one product system (X) is collected, upgraded and used again in another product system (Y). Then, first the boundary has to be determined between the product system producing the waste and the product system using the upgraded waste. For example, collection of waste may be considered as part of product system X and upgrading as part of product system Y. If, in this fashion, there is no process left which contributes to both the waste treatment for product system X and the upgrading for product system Y, no further allocation at all is needed [cf. 6,12]. As long as any such multiple process remains, however, further allocation is necessary. This allocation cannot be based on a common physical parameter, as the multiple outputs are a waste processing *service* and a good. The only possibility in this line of reasoning is to base the allocation on some economic basis, for example the proceeds from the service of waste processing and the sale of the goods produced in the multiple

* Heijungs *et al.* [6] have proposed a method to quantify landfill emissions considering the infinite time-perspective, and degradation and transport processes.

(recycling) process [12,16].

Another method to deal with open-loop recycling has been proposed by Brouwer and Lindeijer [17]. In their method, the primary extraction and production of the materials recycled is partly allocated to product system X and partly to Y according to the loss of quality of the material during the life-cycle of product system X*. In a case study comparing four types of batteries for photovoltaic systems [17] they calculated the results, at the ecoprofile level, for these two different methods for treating open-loop recycling. The differences between the method of Heijungs *et al.* and that of Brouwer and Lindeijer amounted to about 90% for some problem types. This range is merely indicative, as it is based on the result of only one case study and some choices can be made differently within the two methods (e.g. the precise definition or the equation according to which the loss of quality is accounted for), which can influence this range. Choosing the most appropriate allocation method is difficult because, as Brouwer and Lindeijer state, the method described in Heijungs *et al.* rewards the use of recycled material while the method described in Brouwer and Lindeijer rewards the production of recycled material. The two methods are thus complementary, since both activities are needed to achieve a certain level of recycling. However, the goal of allocation methods for open-loop recycling should not be to stimulate recycling, but (also) to indicate whether recycling is environmentally worthwhile in the case studied. Recycling in itself is not always a guarantee for environmental improvement [cf. 18,19].

From the discussion of each of the three types of multiple processes it appears that there is still quite some divergence possible in allocation methods and results, although the basic line of reasoning is quite similar. A stepwise procedure to be followed in each of these three allocation cases (or defined for each type of allocation) giving the different methodological options for each step could structure discussions about allocation. A first proposal for such a procedure was recently made by Huppes [12] and largely agreed upon at a specialized workshop on allocation problems in LCA [20].

6.4.1.3 *Cut-off of processes*

The final step in determining the process tree consists in making choices for limiting the product system to those processes that make a relevant contribution to some environmental input or output. The near-infinite regression of processes needed to produce inputs for other processes should be cut off at the point at which the contribution of another step in the process tree becomes insignificant for the product system studied. However, one can only determine whether the contribution of a process is significant in a given process tree if one knows the environmental impact of that process. Thus, in theory it is very difficult, if not impossible, to make a well considered cut-off.

As discussed in Chapter 2, deliberate cut-offs are often made by including the functioning of capital goods but excluding their production. However, if in a case study on car transportation, for example, the car itself is considered a capital good and if its production is not included in the analysis, a significant error is sure to result. Moll reports in his thesis [21] that in the life-cycle

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* For a more extensive description of this method and the differences with the method described above, see Brouwer & Lindeijer [17]. One of the objections raised against Brouwer & Lindeijer's method is that besides allocating primary extraction and production of the materials recycled in A, disposal of the recycled materials in B should be allocated to both A and B. This makes the allocation procedure more complex, of course.

of car transport, 68% of the total energy requirements is due to actual driving, and 32% to the production of capital goods (the car itself, roads and garages) and maintenance (spare parts and repairs).

Forced cut-offs are usually made because of a lack of data or appropriate calculation methods. With respect to the latter, networks of processes are often cut off to pseudo-linear sequences, as is the case in the study by Habersatter and Widmer [2]. In Chapter 2 it was noted that Boustead [18] calculated that in the case of electricity production the exclusion of the production of capital goods and the simplification of networks to pseudo-linear sequences may introduce an error of approximately 10% with respect to overall energy efficiency, with associated effects. These kinds of errors can easily be avoided today since proper calculation methods are now available [6,18,22].

Experiences from case studies can also be used to make well-considered cut-offs. For example, from a number of case studies on light bulbs [23] it is known that electricity consumption during use is the determining parameter in an LCA of light bulb systems. This result can be used to perform a quick screening LCA of new bulbs, although one should always remain alert to the possibility of other negative environmental effects of the new bulb system [cf. 24].

Another motive for consciously omitting processes is at work when a difference analysis is made, i.e. when two or more product systems are compared and identical parts left out. In some cases this may represent an option for performing time- and money-efficient LCAs.

6.4.2 Process data

6.4.2.1 *Availability*

With respect to the availability of data, it should be noted that many data are lacking, such as process data for seeds, production of fertilizers and pesticides, for forestry, the production of ingredients and cleaning agents, cooling, dish-washing with respect to the margarine part and a number of waste treatment processes.

As stated earlier, a special case of the boundary definition between environment and product system and the quantification of emission data relates to chemicals which are deliberately applied at some sites in the environment for some specific goal but which can also harm the environment at the intended sites as well as at other (unintended) sites. Cases in point are plant nutrients such as nitrogen, phosphorus and potassium and all kinds of pesticides. In this case study, the focus is on pesticides since they were found to have a decisive influence on the final results, but the discussion below is also largely applicable to plant nutrients. There are two related questions to be answered in an LCA case study for these kinds of chemicals:

- where should the boundary between environment and economy be drawn ?
- how can the relevant emission data be determined ?

Particularly in the case of pesticides, boundary definition is not a clear-cut issue [25]. It can be argued that the boundary should be defined at the interface of the agricultural plot and an adjacent ditch or plot, but it can also be argued that any spillage on the plot (as opposed to intended use on the crop) should also be considered as an emission to the soil.

Besides defining the boundary, it must also be determined which fractions of the overall pesticide emission are emitted to water, to soil and to air. Some average figures for these fractions are available from a background document of the Dutch long-range plan on crop protection [26]

and these are used here*. These figures are based on the implicit assumption that the fraction impinging on the crops and remaining there (thus not evaporating, for example) does not result in an emission to the environment. Every molecule of the pesticide remaining on the plant is thus assumed to degrade to harmless metabolites. For some chemicals this assumption may be valid, but a significant number of pesticides are highly persistent and will eventually end up as an emission to the environment, e.g. during washing of the crop before consumption or processing.

In the previous study on margarines by Vis *et al.* [1] the entire amount of pesticides applied was assumed to be emitted to water. Within the framework of the above discussion, this seems to be an extreme option. To investigate the possible range in the final results of this specific case study resulting from different assumptions concerning pesticide emissions, a second option is here defined: 45% of each pesticide applied is used effectively on the crop and completely degraded to harmless metabolites, 20% is emitted to the air (blown away during application, evaporation), 2% is emitted to water (run-off etc.) and 33% is emitted to the soil. These general data have been adapted data from [26]. In this second option, wastage of pesticides on the agricultural plot itself is considered to be an emission to the soil. In a sensitivity analysis, the influence of these two extreme options on the final case study results have been calculated.

6.4.2.2 *Representativeness and quality*

The data sources used in this study have been reported in the introduction: Vis *et al.* [1] for data on the margarines, Habersatter and Widmer [2] and Lundholm and Sundström [3] for data on packaging, energy and electricity generation, transport and waste incineration, Hoogenkamp [4] for data on P_2O_5 production and Sundqvist *et al.* [5] for data on landfill. Methods for a comprehensive assessment of the representativeness and quality of data have not been elaborated [27]. Some initial suggestions have been made, *inter alia*, by Heijungs *et al.* [6]. It would go too far to discuss these suggestions in detail for all the data sources used in this study. However, an example may serve to illustrate that aspects of representativeness and quality may have a considerable influence on the final results of an LCA. The example concerns the discussion about the choice of the most representative process in relation to the goal of the study. The goal of this case study for Van den Bergh Nederland was to obtain an insight into the environmental aspects of the life-cycle of its margarines. Thus, as far as possible, company-specific data were gathered. However, this was not done for packaging materials, since data on this aspect were lacking. Thus, data from Habersatter and Widmer [2] were taken, for example, for the production of cardboard used in the transport packaging of the margarines. Tillman *et al.* [28] also gathered data for this process. Despite the fact that the production of pulp needed for cardboard production was assumed to be situated in the same country in both studies, Ekvall [29] found - in a study comparing these two data sets - that the differences in some data categories, after correction for different material compositions, system boundaries and certain other different assumptions, could still be more than a factor 10.

Except for differences in the representativeness of these two data sets, these differences

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* Of course, the exact numbers depend strongly on the specific chemical used, how it is applied (spraying by tractor or by aircaft) and the weather conditions during application.

may also be partly due to differences in data quality*. The question can be posed whether the two data sets are correct and complete. Mass and energy balances may be useful in verifying this aspect. For example, although Habersatter and Widmer state in their report that they have applied mass and energy balances in their data gathering, Ayres [30] recently criticized their work for the fact that some mass and energy balances appeared to be incorrect. In the same paper he stressed that relevant techniques have been available for years already and that LCA practitioners and developers should incorporate these techniques in their everyday work in order to improve the scientific quality and credibility of their work. For chemical elements, this critique is appropriate for almost every LCA report produced to date. However, to draw up a correct mass balance for every individual chemical element involved in a process is no easy job. Nevertheless, it is very important that LCA developers and practitioners learn from experiences in other fields of science, and if in the case of mass and energy balances useful techniques are available, one should of course utilize them.

This said however, mass and energy balances cannot be the sole means used to verify the correctness and completeness of the quality of process data; one also needs some "Fingerspitzengefühl" for this purpose. For example, the pesticide data used in this study cannot be correct, although the mass balance is. They cannot be correct, because if one compares the individual pesticide emissions due to annual production of margarines by the Van den Bergh Nederland company with estimated world emissions of these pesticides based on data reported by Coppoolse and Kersten [31] and an upscaling method described in this chapter (paragraph on normalization), some emissions due to the margarines appear to exceed the estimated world emissions by a factor 9. Set off against the (very small) share of most other types of emissions in world emissions, the average error in pesticide emission data may easily be as high as a factor 1000† ! It will be clear that these errors dominate the results and cannot be traced by mass balance techniques, but only by sensible thinking about the data, as exemplified in this paragraph‡.

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* The differences are not due to any substantial differences in the age of the data used in the two studies. Although this was the main cause of differences in the case study on milk packaging discussed in Chapter 2, this is not true here.

† One of the causes of this error may be that the method of estimating the pesticide emissions for use in this case study differs from the method used in Coppoolse and Kersten [31]. The pesticide emission data gathered for this case study have been based on the assumption that 100% of the amount applied is eventually emitted and on an average chemical composition of these amounts, since specific data on the individual pesticides applied were lacking ! The pesticide emissions reported by Coppoolse and Kersten were based on the distribution reported in the Dutch longe-range plan on crop protection [26] (see previous paragraph). Another cause of the error may be that, for a given crop, Dutch pesiticide emissions are probably much lower than those in Third World countries. On closer inspection, therefore, the factor 1000 is built up of an "error" in the data gathering for this case study and of an "error" in the estimate of world pesticide emissions.

‡ Another source of error may be the lack of a standardized format for presenting process data and a standard list of parameters to be measured. With respect to format, for instance, there are processes (such as "Zellstoffherstellung" [2]) that both require and emit SO_2. The first item is an economic input, the second an environmental output. These distinctions between economic and environmental data are not self-evident and can lead to (avoidable) mistakes. The same ambiguity holds for the energy production data presented in the very important work of Habersatter & Widmer [2]. For example, it is not clear whether the input of oil for the extraction of fossil oil is processed oil, and thus an economic input, or oil as a resource, and thus an environmental input. The fact that there is no standard nomenclature for inputs and outputs can lead to double countings and, in any case, to confusion. For instance, some sources present both emissions of both BOD and COD, although COD also includes BOD. Furthermore, emissions are sometimes expressed in their elementary form and

6.4.3 Inventory table

Based on the starting points and assumptions described above, the inventory tables of the four margarines have been calculated by applying a matrix method as described in Heijungs *et al.* [6], which was implemented in supportive software* at the Centre of Environmental Science of Leiden University (CML). This matrix method can handle networks of processes, as described above, thus avoiding errors due to simplification of such networks.

The inventory tables of the four margarines cannot be shown in the results section. The company is afraid that, although it has been stated clearly that the reliability of some data is debatable (see the discussion on pesticides above), these data could be misinterpreted and used by third parties without clear notification of their status.

6.5 IMPACT ASSESSMENT

6.5.1 Introduction

In this thesis, a methodology for impact assessment has been developed (Chapters 3, 4 and 5), based on the distinction - according to the Code of Practice terminology [7] - between three steps: classification, characterization and valuation. Other methods for performing the impact assessment in a single step have also been discussed. In this case study, a sensitivity analysis has been performed to compare the results of the multi-step methodology developed in this thesis with the results of two single-step methodologies: EPS as described by Steen and Ryding [32] and the Ecopoints method as described by Braunschweig and Müller-Wenk [33]. The data used for performing an impact assessment according to these methodologies are also from these documents†. Below, the multi-step methodology as developed in this thesis is discussed in more detail within the framework of the case study.

6.5.2 Classification

The first step of the multi-step impact assessment is definition of the impact categories. In Chapter 3 of this thesis an extensive list of impact categories that might be included in LCAs was presented. The impact categories relate to the major environmental themes such as global warming, acidification etc., which explains the fact that this type of impact assessment is often referred to as the "Environmental Themes" approach (impact category = environmental theme = problem type).

Since in the life-cycles of margarines, the use of biotic resources and the emissions of ozone depleting gases are zero, these two impact categories are not shown in the results of this

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sometimes in their compound form. For the elements, equivalency factors are often lacking or should be derived as the average of all possible compounds of that particular element. In some cases, for example an emission to water of N or of NO_2^-, this distinction may make an significant difference in an impact assessment. Similar ambiguities exist with respect to the nomenclature of resources. Standardization would certainly help here.

* This software, named SIMA/2, is still under development. In this case study, calculations have been made with SIMA/2 as developed up to March, 1994. It cannot be guaranteed that the software, and thus the results of this case study, are free of error. Neither CML nor its staff can be held liable for any use of the results.

† As mentioned, the inventory tables cannot be shown here. Unfortunately, this means that it is not possible to give the impact assessment data for the different methodologies per inventory item.

case study. Furthermore, data on emissions of radioactive chemicals (e.g. related to nuclear power plants for electricity production), noise and chemicals on the shopfloor were not available, if relevant at all. These impact categories were therefore excluded from this case study. Because data to express land use in an appropriate form (e.g. annual production per process) [cf. 6,34,35] and elaborated equivalency factors were lacking, land use could not be included either. Since energy balances were lacking in the energy data reported by Habersatter and Widmer [2] and have not been made in this limited case study, waste heat was also excluded. Summarizing, in this methodological case study the problem types for which data are available and non-zero and for which methods are available are:

- depletion of abiotic resources
- global warming
- photochemical oxidant formation
- acidification
- human toxicity
- aquatic ecotoxicity
- terrestrial ecotoxicity
- nutrification
- smell
- desiccation*.

6.5.3 Characterization

The second step of the impact assessment is characterization, in which - again according to the Code of Practice [7] - "[...] aggregation of the impacts within the given impact categories takes place. This step should be based on scientific knowledge about environmental processes. There are various approaches to characterization."

In this thesis an approach has been developed which is often referred to as the "Environmental Themes Approach". Characterization according to the environmental themes approach is performed by multiplying the inventory data (extractions and emissions) by so-called equivalency factors and by aggregating the results of these multiplications in one or more effect score(s) per impact category. Equivalency factors have been proposed for all the impact categories included in this case study (see Chapter 3, 4 and 5 of this thesis and Heijungs *et al.* [6]).

The problems related to this type of characterization can be divided into four categories:

1 inadequacy of equivalency factors;
2 lack of data and lack of knowledge;
3 gaps between inventory data and equivalency factors;
4 uncertainties in data.

Definition of a number of equivalency factors is still under development and thus still inadequate.

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* Desiccation is expressed in m^3 of water used. Water use is rarely specified by source, which may give rise to interesting environment-economy boundary discussions. The source may be a river, groundwater or a public water supply system. Rivers and groundwater are environmental sources and thus environmental inputs. Public water supply systems are part of the economy and should be treated as such, i.e. all environmental impacts associated with (waste) water treatment should be allocated to the supply of this kind of water. As far as is known to the author, this distinction has never been made in any LCA study.

For example, the equivalency factors applied in this case study to classify abiotic resources are based on a preliminary method described in Heijungs *et al.* [6]. These equivalency factors are unsatisfactory because they were based on the economic reserves of those particular resources and did not take production levels into account, and were elaborated for only a very few resources. In Chapter 5 of this thesis another method has been proposed for assessing these resources. In a sensitivity analysis, the results of both methods are compared for this case study. Other examples of inadequate equivalency factors are those for nutrification, acidification, ecotoxicity and human toxicity. The factors for the impact categories are incomplete, as an assessment of the fate of the chemicals emitted into the environment is not included. In Chapter 4, a proposal for improving the characterization of ecotoxic and human-toxic chemicals has been developed, but its implementation still involves practical problems. Besides these practical problems, the method proposed in Chapter 4 has been challenged by Assies [36] and defended by Heijungs and Guinée [37].

This brings us to the second problem category: the lack of data and knowledge. For the characterization of ecotoxic and human-toxic emissions according to the method described in Chapter 4 of this thesis, data such as vapour pressure, degradation rates per compartment etc., are needed for each chemical (and in fact also for its metabolites). These data are often lacking because knowledge on these aspects is still limited, which is one reason that this method is not yet sufficiently practical. Also, in current characterization methods for ecotoxic and human toxic emissions, data sets are far from complete due to a lack of knowledge*. How these data gaps are handled constitutes an important issue, but methods have not yet been elaborated for this problem within the framework of LCAs. Probably the best way, for the time being, is to show all data gaps as precisely as possible, e.g. in a matrix table showing, for all inventory data, whether an equivalency factor is present, relevant but missing, or irrelevant. With respect to this case study, it can be stated that most substances for which equivalency factors are lacking are quite harmless (e.g. KCl), some *might* be harmful (e.g. mercaptan) and some are evidently harmful (e.g. PAH and a large number of herbicides). The latter category of harmful chemicals for which equivalency factors are lacking may significantly influence the three toxicity scores†.

This brings us to the third, closely related, problem: gaps between inventory data and equivalency factors. Some chemicals are sometimes measured as group parameters in the inventory analysis, for example PAHs and CFCs, while equivalency factors are only available for individual chemical species, for example anthracene and chrysene, or CFC-11 and CFC-12. Other group parameters that are often encountered in inventories and for which this problem occurs are organic compounds, sulphur compounds, absorbable organic halogens (AOX) and C_xH_y (hydrocarbons). If the individual emission data are not available, as in this case study, the least one can do, of course, is to work with the arithmetic means of the individual species within that group. If, for example, only emission data on "hydrocarbons" are known, group POCPs can be derived as the arithmetic mean of the POCPs of individual hydrocarbons. This approach is of value only if the POCPs of the individual hydrocarbons vary within a "reasonable" range, which was assumed in

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* The discussion about the indirect GWPs for NO_x, NMHC, CH_4 and CO is also an example of this lack of knowledge (cf. Chapter 3 and [6]).

† Due to the fact that the inventory table cannot be shown, the data gaps of this study cannot be presented in any detail.

this case study for C_xH_y. However, arithmetic means, for organic compounds and AOX for example, have little meaning since the variation between the occurrence and/or the equivalency factors of the individual substances is extremely large. In fact, a specification of these group parameters is required for characterization. Even though incomplete, any specification is better than none, however, and this also holds for the hydrocarbons discussion.

A final but not unimportant problem is the uncertainty of some equivalency factors. For example, the global warming potential is defined for different time horizons. It is difficult to determine which time horizon is the best for LCAs (see discussion in [6]) and probably the best thing to do is to calculate the results of a case study for different time horizons and see what the differences are. Such ranges are given in Heijungs *et al.* [6] for global warming potentials (GWPs) and photochemical oxidant creation potentials (POCPs). A sensitivity analysis of the effect of the upper and lower values of these factors on the results of this case study is presented in the next section. For a more structural and comprehensive review of the uncertainties involved in classification and characterization according to environmental themes - and in LCAs in general - see Van den Hout [38]*.

Normalization

It is difficult to interpret the effect scores constituting the environmental profile. This is because the order of magnitude and units of the various effect scores differ. For this reason, it was proposed in Chapter 3, and in the Code of Practice [7], to normalize the effect scores, in other words to divide each effect score by the total magnitude of the effect score for a particular area and a given period of time. Recently, a first elaboration of this step was made [39]. The area considered was the whole world and the period considered one year. The world effect scores for one year were calculated according to the classification/characterization model discussed in Chapter 3 and applied in this case study.

One of the major problems in operationalizing the normalization step of LCAs is to obtain world emission data. World resource extraction data appeared to be quite readily available, as the US Bureau of Mines already publishes such data [40]. Furthermore, for global warming, ozone depletion and acidification, this is a relatively easy job since emission inventories for these problems are (largely) available [41]. For other problem types considered in this case study, such as human toxicity, aquatic and terrestrial ecotoxicity, photochemical oxidant formation, nutrification and smell, the job is much more difficult. For the emissions contributing to these problem types, global inventories are not available. For these emission types, Dutch data from the Dutch Emission Registration database [42,43] have been used and upscaled to world emission data by multiplying the Dutch data by the ratio between the World Gross National Product (GNP) and the Dutch GNP, which is about a factor 100 [44].

The result of this normalization can be viewed as a challenge to experts in the fields of emission inventories and related subjects to develop better data and/or a better upscaling method

* The review of uncertainties by Van den Hout is thorough and interesting, but recommendations for handling these uncertainties are unfortunately largely lacking.

than the one based on the GNP ratio*. In this chapter, the very first elaboration is presented, including the updating which has already taken place [45; see below]†.

Table 2 shows the values of the world annual effect scores of a number of problem types. The values are derived from [45], corrected for some mistakes [pers. comm. with H.A.J. Senhorst, Department of Waterways and Public Works, The Netherlands] and updated and extended, based on additional Dutch emission inventories [31,46].

Table 2: World annual effect scores for a number of problem types (Source: [45,46])

effect score	value
abiotic depletion (yr^{-1})	1.13
global warming ($kg.yr^{-1}$)	3.77×10^{13}
photochemical ozone formation ($kg.yr^{-1}$)	3.74×10^{9}
acidification ($kg.yr^{-1}$)	2.86×10^{11}
human toxicity ($kg.yr^{-1}$)	3.24×10^{11}
aquatic ecotoxicity ($m^3.yr^{-1}$)	9.08×10^{14}
terrestrial ecotoxicity ($kg.yr^{-1}$)	1.26×10^{15}
nutrification ($kg.yr^{-1}$)	8.87×10^{10}
smell ($m^3.yr^{-1}$)	6.28×10^{17}
desiccation ($m^3.yr^{-1}$)	3.24×10^{12}

Normalization offers a wide range of opportunities for presenting results. One presentation method, suggested by Wenzel and Hauschild [47], is to divide the world annual effect scores by the world's total human population, which results in the average contribution per world citizen per year to the environmental problems considered‡. If one chooses to take annual consumer consumption as the functional unit of the product system, as is done in this case study, and the normalized effect scores of that product system are divided by the average annual contribution per world citizen, the result can be expressed as the percentage that the functional unit studied contributes to the total annual contribution to a particular environmental problem of one average

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* The status of this elaboration is very preliminary and the reliability of the figures produced is doubtful. Reactions from the field have been similar: LCA practitioners consider such normalization to be a very useful step, but elaboration to date contains many errors. However, the goal of this initial elaboration was to get the job started and not to produce final results. Calculation of world effect scores, which could serve a much broader purpose than merely within LCA, for example as indicators of the "world's environmental state" at a certain time, is not such an easy job because emission inventories need to be linked to equivalency factors with all the nomenclature and interpretation problems implied.

† In a document containing data on more than 600 different chemicals which was written within a short time, all kinds of errors are possible: typing errors, missing equivalency factors, misinterpretation of emission inventory data, etc. [see 45]. Quite a number of errors have already been traced in the original normalization document, but there are probably still a number of errors. Further peer review remains necessary !

‡ The assumption is then made that all environmental problems are directly or indirectly related to consumer activities, which is essentially true.

world citizen. A formula for an effect score expressed in kg is:

$$\textit{annual world citizen contribution}_i \text{ (kg/yr/person)} = \frac{\textit{world annual effect score}_i \text{ (kg/yr)}}{\textit{world human population (person)}} \quad (3)$$

and

$$\textit{product citizen contribution}_i \text{ (\%)} = \frac{\textit{product annual effect score}_i \text{ (kg/yr/person)}}{\textit{annual world citizen contribution}_i \text{ (kg/yr/person)}} \times 100\% \quad (4)$$

Another way to present the results of a case study is to express them in terms of a functional unit of a reference product generally known to be contributing to a particular problem type*.

If one adds a valuation step (see below) weighting the different problem types into one overall environmental index, this index can be expressed in a similar way to that discussed for the effect scores.

6.5.4 Valuation

The last step of the impact assessment component is valuation, which involves weighting the different (normalized) effect scores against one another in order to make a final appraisal in terms of one aggregated index. It was argued above (Chapter 3) that a quantitative multi-criteria analysis based on a standard list of weighting factors is preferable for LCA. The question, however, is how to these weighting factors should be determined. In Heijungs *et al.* [6] four methods for this purpose were discussed:

- weighting using monetary methods;
- weighting by societal preferences;
- application of sustainability standards;
- modelling of the eventual effects.

There are several examples of monetary weighting systems known within the framework of LCA: the EPS system of Steen and Ryding [32] in which the concept of willingness-to-pay is applied; the Tellus method, as described in their 1991 report "Disposal Cost Fee Study" [48], in which emissions of gases are aggregated based on the cost of preventing environmental effects, and emissions of metals are aggregated based on the cost of repairing the environmental effects of these emissions; and the DESC method, as described by Krozer [49], in which emissions are aggregated based on the cost of reducing them to an acceptable level. These methods are all single-step impact assessment methods. No example is yet known of a monetary weighting method within a multi-step impact assessment.

In weighting by societal preferences the opinion of an individual or a group of individuals might be surveyed [6]. This method has been applied once [50] within the framework of LCA, and was recently further developed by Kortman *et al.* [51]. One of the main conclusions of the study of Kortman *et al.*, which is supported and illustrated in a paper by Heijungs [52], was that the precise questions asked to determine the individual preferences are crucial.

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* Such a method was applied by G. Parker in his presentation of a case study on toothpaste packaging at the SETAC-Europe Symposium for Case Studies on December 7, 1993. He presented the improvement of a toothpaste package in terms of having an X watt light bulb burn for Y hours in Z households.

In weighting by sustainability standards, a weighting factor might be defined based on an "acceptable" level of the effect [6]. The term "acceptable" indicates that a sustainability standard will always include a normative assessment of the level deemed to be tolerable. For most problem types, only zero extraction and emission scenarios will (eventually) lead to zero effect levels. Although zero effect levels may be environmentally preferable, they are not very realistic. Initial efforts to elaborate this approach have been made by Adriaanse [53], Kortman *et al.* [51] and Tukker [54]. This approach can be based on present standards (indicating the effect level to be aimed at in the short term) or target standards (indicating the effect level to be aimed at in the longer term) determined by governments and, of course, all kinds of standards intermediate between the two. The target standard will, however, give the best indication of a sustainable effect level and thus seems preferable in this approach.

Another approach would be to complete the last part of the effect chain and model the ultimate effects. Environmental effects would then be expressed in terms of such target variables as human health and welfare and intrinsic nature values. If it proves possible to model this last part of the effect chain on the basis of environmental science, this valuation part of the impact assessment should be shifted to the classification and characterization. However, a weighting of the various target variables (or "areas of protection", cf. [7]) against one another would still be necessary in many cases and for this weighting one of the aforementioned methods would again be necessary.

As suggested by Kortman *et al.* [51], by Lindeijer at the SETAC-Europe congress in Brussels (April 11-14, 1994) and by Heijungs [55], a combination of two or more of these four methods is also feasible and perhaps even preferable. For example, weighting factors could first be based on sustainability standards, which could serve as input information for a weighting by societal preferences.

This kind of combined weighting approach has been taken here. Thus, two partial weighting factors which should be multiplied together and with the normalized effect scores can be distinguished:

a) a sustainability factor, indicating the distance between the current and sustainable level of a problem; this factor is based on both an objectified scientific environmental assessment of environmental effects and a normative societal valuation of the feasibility of that particular sustainable level;

b) an inter-effect factor, indicating the relative seriousness of the different effects if they actually occur, e.g. ten square metres of acidified forest versus one case of human cancer; this factor is based exclusively on a normative (social) valuation.

In this case study the sustainability factor will be discussed in more detail. The inter-effect factor will not be elaborated any further but is assumed to be unity for all problems [cf. 51], not because this factor is considered less important, on the contrary, but rather because its elaboration constitutes a study in its own right and is not within the scope of this limited case study*.

In elaborating the sustainability factor, three questions must be addressed:

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* As discussed previously, given an accepted valuation method, a world environmental index might also be calculated, indicating the global state of the environment at a given time. Such an index would be calculated by multiplying the world effect scores by the matching sustainability factors and the matching inter-effect factors and adding the results of these multiplications.

1 which parameters are relevant ?
2 which data sources are used ?
3 how are these parameters weighted in relation to one other, or, in other words, which mathematical equation is to be applied ?

It is argued here, and also by others [cf. 51], that the normalized effect scores should be weighted against the current level of the problem N, as quantified in the normalization, and against the long term target standard D^*.

The next question, which data sources should be used to quantify N and D for the various impact categories, is far more difficult, as there is no generally accepted set of target standards available for the different problem types. To prevent lengthy discussions on this topic within the framework of this case study, standards from different sources have here been taken, quite arbitrarily, without suggesting that these are the correct ones. These standards and sources are shown in Table 3.

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* This weighting against N and D shoud not be confused with the weighting against N in the normalization, because they involve two different and complementary questions. In the normalization, the contribution of the effect scores of the functional unit studied to N (the current level of the problems) is calculated to gain an impression of the contribution of that functional unit to the total extent of the various problems. In the valuation, the distance between D (the sustainable level of the problems) and N is determined through the sustainability factor in order to gain an impression of the seriousness of that problem.

Table 3: "Assumed target standards" per problem type

problem type	D	assumption; source
abiotic depletion (yr^{-1})	0.56	50% reduction assumed as target values are lacking
global warming (kg/yr)	2.28×10^{13}	[53]
ph. ozone formation (kg/yr)	1.12×10^{9}	70% reduction [56]
acidification (kg/yr)	4.83×10^{9}	Dutch most rigorous target value [56] × surface of The Netherlands × 100 (GNP-based factor discussed above)
human toxicity (kg/yr)	3.52×10^{11}	world's human body weight as calculated in [6]
aquatic ecotoxicity (m^3/yr)	1.26×10^{14}	world's volume of surface fresh water [57]
terrestrial ecotoxicity (kg/yr)	7.32×10^{13}	world's mass of (ice-free) soil assuming average relevant depth of 10 cm [57]
nutrification (kg/yr)	5.60×10^{9}	based on Dutch target values for N and P × 100 (GNP-based factor discussed above) × matching equivalency factors [58]
smell (m^3/yr)	3.00×10^{18}	world's volume of air [57,59]
desiccation (m^3/yr)	4.07×10^{13}*	world's annual internal renewable resource [46]

The last and key question addressed here is to find expressions indicating the gap between current and target problem levels and consider the different results they lead to. Two expressions discussed by Kortman *et al.* [51] and Heijungs [55] are:

$$\textit{sustainability factor}_i = \frac{N_i - D_i}{D_i} \text{ for } N_i > D_i \qquad (5)$$
$$\textit{sustainability factor}_i = 0 \text{ for } N_i \leq D_i$$

$$\textit{sustainability factor}_i = \frac{N_i}{D_i} \qquad (6)$$

where N_i is the world annual effect score for problem i and D_i is the target level of problem i

* At the global level, smell and desiccation are no problem at all, as can be concluded from their D values. However, desiccation, for example, is certainly a regional problem, as can be concluded from the values of N_{desicc} and D_{desicc} for the Netherlands, which are 14.47×10^{9} and 10×10^{9} m^3 water per year, respectively [46]. The conclusion from this example is that it may be useless to include regional problems in a global generic classification [see Chapter 3] or, in other words, that for regional problems a region-specific impact assessment should be developed. The same conclusion might be expected for nutrification, but the figures for this problem type are somewhat misleading as they are based on the Dutch situation and upgraded to world annual figures, as global data - which are available for desiccation in terms of annual withdrawals and annual internal renewable stocks - are lacking here.

expressed according to the same classification/characterization model as N_i.

The effect scores (ES_i) normalized by N_i can now be valuated and aggregated in two ways into an environmental index:

$$environmental\ index\ (yr) = \sum_i \left[\frac{ES_i}{N_i} \times \frac{N_i - D_i}{D_i} \times I_i \right] \tag{7}$$

$$environmental\ index\ (yr) = \sum_i \left[\frac{ES_i}{N_i} \times \frac{N_i}{D_i} \times I_i \right] \tag{8}$$

where I_i is the inter-effect factor which is, as discussed, provisionally assumed to be unity. Note that in Eqn. (8) N becomes an irrelevant parameter and normalization is no longer necessary for such a valuation.

It might also be argued that normalization should be performed by dividing the effect scores by the sustainable problem levels (D_i). Then, two other possibilities are*:

$$environmental\ index\ (yr) = \sum_i \left[\frac{ES_i}{D_i} \times \frac{N_i - D_i}{D_i} \times I_i \right] \tag{9}$$

$$environmental\ index\ (yr) = \sum_i \left[\frac{ES_i}{D_i} \times \frac{N_i}{D_i} \times I_i \right] \tag{10}$$

The various equations should be checked as to whether they satisfy certain basic criteria. For example, Heijungs [55] observed that all equations including $N_i - D_i$ are debatable. For example, $N_i - D_i$ becomes negative if $N_i < D_i$, which might encourage emission of as much as possible of substances contributing to problem i for which $N_i < D_i$, since this will lower the environmental index†. Even if one makes the modification embodied in Eqn. (5), the equation would still not be satisfactory, because it would make no difference whether one contributes to this problem to some extent or not at all, which will lead to no more pollution reduction than strictly necessary. This focuses the discussion on two equations: (8) and (10). These are the options applied in this case study and for which a sensitivity analyses has been performed.

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* In fact there are three other possibilities, but one:

$$environmental\ index_{functional\ unit} = \sum_i \left[\frac{ES_i}{D_i} \times \frac{N_i - D_i}{N_i} \times I_i \right]$$

turns out to be similar to Eqn. (7).

† This example shows that the search for a proper equation involves obvious normative considerations.

6.6 RESULTS

In this section the results of the case study are presented at the level of the ecoprofiles, the normalized ecoprofiles and the environmental indices for the four margarines. As discussed above, some choices had to be made during the calculation of these results. For this, the following list of default choices was drawn up:

- allocation of coproduction on a mass/energy basis, according to the allocation factors presented in Table 1 (second of four columns);
- assumption that the entire quantity of pesticides applied to the crops is emitted to water;
- characterization of the inventory data according to the method described in Chapter 3 of this thesis and the equivalency factors reported by Heijungs *et al.* [6]; assumption of a GWP time frame of 100 years; and assumption that the effect score for photochemical oxidant formation is based on average POCP values;
- calculation of the environmental indices according to Eqn. (10).

In the next section on sensitivity analyses, these default values are chosen differently in order to investigate the extent to which these choices may affect the results of the case study.

The ecoprofiles of the four margarines, calculated according to these initial choices, are shown in Table 4.

Table 4: Ecoprofiles of four margarines

effect score	A	B	C	D
abiotic depletion (dim.less.)	2.2e-14	1.9e-14	3.6e-14	3.1e-14
global warming (kg)	7.3	6.2	6.4	5.0
photochemical oxidant formation (kg)	0.024	0.024	0.027	0.027
acidification (kg)	0.14	0.098	0.090	0.057
human toxicity (kg)	0.17	0.13	0.13	0.082
aquatic ecotoxicity (x 10^6 m^3)	0.92	1.2	2.2	1.5
terrestrial ecotoxicity (kg)	120	70	104	73
nutrification (kg)	0.25	0.17	0.28	0.20
smell (x 10^3 m^3)	3.6	3.5	1.1	1.5
desiccation (m^3)	0.022	0.015	0.014	0.012

The various effect scores point in different directions for the four margarines. For example, margarine A appears to be the best in terms of aquatic ecotoxicity, margarine B in terms of abiotic depletion, terrestrial ecotoxicity and nutrification, margarines A and B in terms of photochemical ozone creation, margarine C in terms of smell, and margarine D in terms of global warming, acidification, human toxicity and desiccation.

To view the values for the effect scores presented in Table 4 in the perspective of the global annual extent of the various problem types, a normalization has been performed, the results of which are shown in Table 5.

Table 5: Normalized ecoprofiles of four margarines (x 10^{-12} year)

effect score	A		B		C		D	
abiotic depletion	0.019	(1)	0.017	(0.90)	0.032	(1.7)	0.027	(1.43)
global warming	0.19	(1)	0.16	(0.85)	0.17	(0.88)	0.13	(0.69)
photoch. ox. formation	6.4	(1)	6.4	(1.0)	7.2	(1.1)	7.2	(1.1)
acidification	0.50	(1)	0.34	(0.69)	0.31	(0.63)	0.20	(0.40)
human toxicity	0.54	(1)	0.40	(0.74)	0.40	(0.75)	0.25	(0.47)
aquatic ecotoxicity	$1.0x10^3$	(1)	$1.4x10^3$	(1.4)	$2.5x10^3$	(2.4)	$1.6x10^3$	(1.6)
terrestrial ecotoxicity	0.095	(1)	0.055	(0.58)	0.083	(0.87)	0.058	(0.61)
nutrification	2.8	(1)	1.9	(0.69)	3.2	(1.1)	2.2	(0.80)
smell	0.0057	(1)	0.0056	(0.98)	0.0018	(0.31)	0.0024	(0.42)
desiccation	0.0068	(1)	0.0046	(0.68)	0.0042	(0.62)	0.0038	(0.55)

(..) = value relative to corresponding value of margarine A

This table shows that the contribution of the margarine systems to aquatic ecotoxicity is extremely high, which is mainly due to the erroneously high pesticide emissions, as will be obvious from the above discussions. On the other hand, there may also be errors in the values of the world annual effect scores, as the emission inventories used to calculate them are incomplete and the upscaling method applied is a rough one. However, the influence of this error is probably much lower than that of the erroneous pesticide emissions, which is estimated to be about a factor 1000. If normalized effect scores are calculated for the annual production of margarines by Van den Bergh Nederland, it appears that the world annual aquatic ecotoxicity score is almost parallelled by the contribution of Van den Bergh Nederland annual margarine production alone, which is of course impossible. In this way, normalization may (in future) also serve as a check on the correctness of process data. The error was estimated earlier to be a factor 1000. The normalization results presented in Table 5 seem to confirm this estimate.

As discussed above, other presentations of results based on an alternative method of normalization are possible. For example, the results of this case study might be expressed as the relative share of the annual consumption of margarine in the total annual contribution to the various impact categories of an average world citizen according to Eqn. (4). The results of this alternative normalization are presented in Table 6.

Table 6: Relative share (%) of the annual average consumption of margarine per citizen in the total annual contribution to the various impact categories of an average world citizen (assuming the world human population is 5×10^9 people)

effect score	A	B	C	D
abiotic depletion	0.010%	0.0085%	0.016%	0.014%
global warming	0.10%	0.082%	0.084%	0.067%
photochemical oxidant formation	3.2%	3.2%	3.6%	3.6%
acidification	0.25%	0.17%	0.16%	0.10%
human toxicity	0.27%	0.20%	0.20%	0.13%
aquatic ecotoxicity	506%	687%	1234%	818%
terrestrial ecotoxicity	0.048%	0.028%	0.041%	0.029%
nutrification	1.4%	0.97%	1.6%	1.1%
smell	0.003%	0.003%	0.001%	0.001%
desiccation	0.003%	0.002%	0.002%	0.002%

On the one hand, Table 6 makes clear what we are talking about. On the other hand, it might lead to an attitude of apathy. Seen in a broader perspective, any product group is only responsible for an extremely small part of overall environmental problems. However, inspired by the belief that we have to improve the environmental impact of the entire economic system, every bit helps, and the same holds for analyzing and improving margarine systems.

Applying the data for *N*, *D* and *ES* as given in Tables 2, 3 and 4, respecively, the environmental indices for margarines A, B, C and D can be calculated according to Eqn. (10).

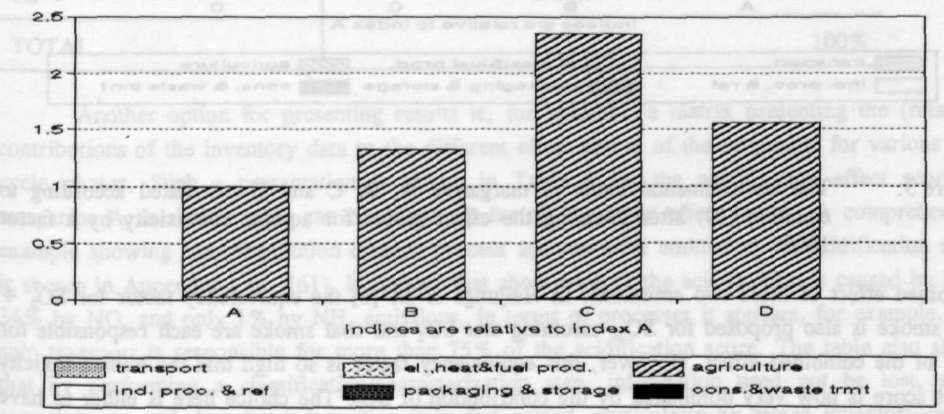

Figure 2: The environmental index of margarines A, B, C and D calculated according to Eqn. (10). Note that all life-cycle phases but agriculture (pesticides) are negligible!

The results are shown graphically in Figure 2. The order of environmental preference according to Eqn. (10) appears to be A-B-D-C. From the bar graph it can be seen that the indices are completely dominated by the agriculture phase*, which is obviously due to the erroneously high pesticide emissions. To check whether this result changes if it is not completely dominated by the (normalized) effect score for aquatic ecotoxicity, the indices for the four margarines according to the two equations have been recalculated, lowering the effect scores for aquatic toxicity by a factor 1000. The results are shown in Figure 3. The results are completely different and other phases of the life-cycle than agriculture then also appear to be important. The order of environmental preference is now D-B-C-A, with A changing from the most preferred to the least preferred position (!).

All kinds of mixes of presentations of results are possible, which can give new information. An example is Table 7, which shows the relative contributions to the effect score for human toxicity of various chemicals emissions. This table shows that SO_2 is responsible for 63% of the total effect score for human toxicity. This high contribution is remarkable, but can nonetheless be explained. Emissions of SO_2 are known to contribute to human toxicity, but there is no specific equivalency factor available. There is such a factor for SO_2 + coal smoke, which is related to the

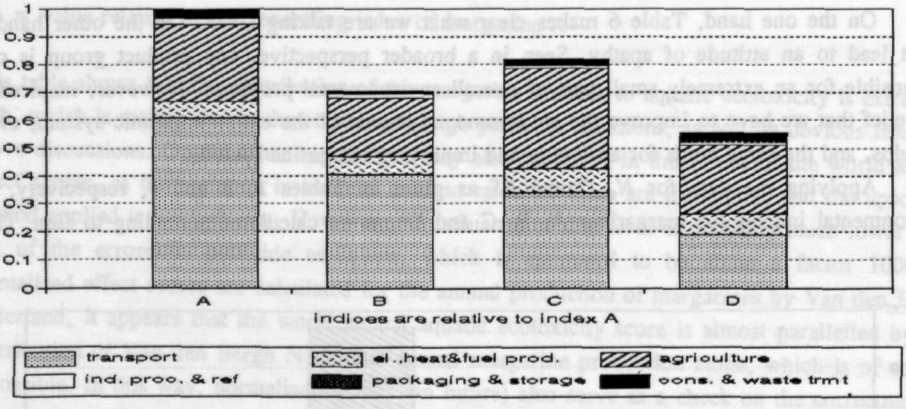

Figure 3: The environmental index of margarine A, B, C and D calculated according to equation (10) after lowering the effect scores for aquatic ecotoxicity by a factor 1000

combined effect of these two emissions. In Heijungs *et al.* [6] the equivalency factor for SO_2 + coal smoke is also proposed for SO_2, assuming that SO_2 and coal smoke are each responsible for 50% of the combined effect. However, this equivalency factor is so high that the human toxicity effect score is now very dominated by the contribution of SO_2. The choice here is either to have no equivalency factor or to have a value that is potentially too high. The second option has been

* The definition of life-cycle phases is an entirely subjective issue. The definition of the various phases used in this particular study is given in Appendix 1.

chosen, which will always remain debatable.

Table 7: Percentage contributions to the human toxicity score by chemicals emitted during the life-cycle of margarine A

chemical emitted	relative contribution to human toxicity of margarine A	
EMISSIONS TO AIR		95%
CO	0.55%	
lead	0.33%	
NO_x	31%	
SO_2	63%	
EMISSIONS TO WATER		4.6%
PCBs	0.15%	
insecticides	3.5%	
herbicides	0.61%	
sulphite	0.39%	
EMISSIONS TO SOIL		0.01%
Cd	0.01%	
TOTAL		100%

Another option for presenting results is, for example, a matrix presenting the (relative) contributions of the inventory data to the different effect scores of the ecoprofile for various life-cycle phases. Such a presentation is shown in Table 8 for the acidification effect score of margarine A; a similar table can be drawn up for all the other effect scores. A comprehensive example showing the contribution of each process and chemical emitted to the acidification score is shown in Appendix 1 (p. 161). It appears that about 64% of the acidification is caused by SO_2, 34% by NO_x and only 1% by NH_3 emissions. In terms of processes it appears, for example, that ship transport is responsible for more than 75% of the acidification score. The table also shows that by performing a classification/characterization step, information need not be lost, as is sometimes argued, but that information is in fact gained, particularly by mixed inventory-impact assessment presentations, as shown in Tables 7 and 8 [60].

Table 8: Relative contributions of inventory items to effect score for acidification per life-cycle phase

substance	T	E	A	I	P	C	W	Total
NO_x	30%	2.0%	0.02%	0%	0.04%	0%	2.5%	34%
SO_2	57%	6.7%	0%	0.22%	0.10%	0%	0.68%	64%
NH_3	0%	0.04%	0%	0.97%	0%	0%	0%	1.0%
HCl	0%	0%	0%	0%	0%	0%	0.20%	0.20%
Total contr.	87%	8.7%	0.02%	1.2%	0.14%	0%	3.4%	100%

T=transport; E=electricity, heat & fuel production; A=agriculture; I=industrial processing & refining; P=packaging & storage; C=consumption; W=waste processing; Total contr.=contribution to acidification score

Similar tables can be developed at the level of an environmental index, showing, for each process, the contributions of the various inventory data and effect scores to the environmental index. An example of such a simplified matrix for margarine A is shown in table 9*. This example can be kept this simple because in the current impact assessment of this case study the results are completely dominated (97%) by rape seed growing (because of the erroneously high pesticide emissions) and to a far lesser extent by ship transport (2.5%). If this had not been the case, it would not have been possible to show the complete matrix, as this one is far too large (e.g. a square matrix of dimension 100).

* The environmental index is calculated according to Eqn. (10), presented above, and the values for N_i and D_i as presented in Tables 6 and 12, assuming the inter-effect factor to be 1 for all problem types.

Table 9: Contributions of some emissions and effect scores to the environmental index of margarine A for two dominating processes

emissions and effect scores	rape seed growing	ship transport
aquatic ecotoxicity effect score	95%	0%
oil	0%	0%
phenol	0%	0%
insecticides	95%	0%
herbicides	0.05%	0%
terrestrial ecotoxicity effect score	0.04%	0%
Cd	0.01%	0%
zinc	0.03%	0%
acidification effect score	0%	2.5%
NO_x	0%	0.71%
SO_2	0%	1.8%
nutrification effect score	1.1%	0.03%
NO_x	0%	0.03%
nitrogen	0.51%	0%
phosphorus pentoxide	0.58%	0%
photochem. ox. form. effect score	0%	0.02%
C_xH_y	0%	0.02%
TOTAL	97%	2.5%

Note: In this table the effect scores for aquatic ecotoxicity have no longer been lowered by a factor 1000 !

These kinds of presentation can be very useful for improvement assessment. They enable one to trace the main origins of the different effect scores or the environmental index, which may in turn allow options for improving the environmental performance of the product system studied to be identified. As stated above, improvement assessment is not further elaborated here, due to the limited scope of this case study.

Table 9 also reveals that the contribution to acidification by emissions of NO_x and SO_2 is due to ship transport. Although these emissions also dominate the human toxicity score, as shown in Table 7, in this case the human toxicity score does not significantly contribute to the environmental index and it is not therefore shown in Table 9. However, ship transport is mainly at sea, so that the bulk of these emissions occur at sea, where it can be argued that acidification and

human toxicity are not as severe problems as they are on continental land masses. Potting [61] discussed this problem and suggested to make the impact assessment more "area dependent" by indicating in generic terms the place of the emission, e.g. at sea or on continental land masses. Although her proposals have still to be discussed in detail and elaborated further*, this suggestion by Potting seems a promising one for a feasible spatial differentiation of the impact assessment in LCA - also in practical sense (cf. Chapter 3 of this thesis).

6.7 SENSITIVITY ANALYSES

Analysis of the sensitivity of the results to a number of choices forms the subject of this section. These sensitivity analyses have not been performed to enable a final appraisal of the status of the results of this specific case study to be made, but rather to evaluate the different methodological issues which need further attention. Sensitivity analyses have been performed, comparing results at the level of the environmental index, for the following topics:

- allocation of coproduction;
- estimation of pesticide emissions;
- application of a different method for characterizing abiotic resources, and
- the equation used to calculate the sustainability factor.

As shown in the previous section, on results, it is to be expected in advance that the erroneously high pesticide emissions dominate the results to such an extent that sensitivities due to other choices, methods and data than those related to these pesticide emissions are unlikely to be detected. Therefore, all sensitivity analyses have been performed on the basis of effect scores for aquatic toxicity that have been lowered by a factor 1000†. The factor 1000 should be considered as a rough estimate of the error in the pesticide emissions.

6.7.1 Allocation of coproduction: mass/energy versus economic basis

The environmental indices of the margarines A, B, C and D have been recalculated, allocating coproduction processes on an economic basis according to the allocation factors presented in Table 1 (third of four columns). The results are not shown here because it appeared that the sequence of environmental preference of the margarines at the level of environmental indices remained exactly the same. However, the results at the level of individual effect scores do change. These results are presented in Table 10. This table shows the relative differences between the effect scores of the four margarines as allocated on an economic and on a mass/energy basis. These differences may be as high as 70% or more, but are similar for all four margarines, which explains why the relative environmental indices do not change. The increase is equal for all four margarines, because their process trees are all quite similar and as far as they include different processes, these processes are again quite similar, which is not particularly surprising since the margarines are

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* For example, transport of these emissions to the continents, resulting in continental acidification, should then also be taken into account; how this might be done is still unclear, however.

† Thus, Tables 4, 5 and 6, for example, are still valid, but the effect scores for aquatic toxicity in these tables should be lowered by a factor 1000.

produced by the same manufacturer. This means that although the final appraisal of the four margarines is not influenced by this alternative allocation method, the absolute values will influence the assessment of other product systems that use the coproducts as an economic input.

Table 10: Relative differences between effect scores of four margarines with coproduction allocated on an economic basis instead of a mass/energy basis:

$$difference_i\ (\%) = \frac{effect\ score_{i,economic} - effect\ score_{i,mass/energy}}{effect\ score_{i,mass/energy}} \times 100\% \qquad (11)$$

effect score	A	B	C	D
abiotic depletion	7.1%	11%	3.6%	2.4%
global warming	7.8%	11%	7.7%	5.1%
photochemical oxidant formation	9.2%	18%	5.5%	3.6%
acidification	8.5%	12%	12%	8.5%
human toxicity	11%	19%	24%	20%
aquatic ecotoxicity	72%	69%	74%	69%
terrestrial ecotoxicity	61%	64%	72%	68%
nutrification	67%	66%	72%	66%
smell	0.51%	1.2%	3.6%	1.1%
desiccation	30%	21%	9.6%	3.3%

6.7.2 Emissions of pesticides

To assess the possible range in the final results of this specific case study resulting from different assumptions with respect to pesticide emissions, the results have been recalculated assuming that 45% of each pesticide applied is used effectively on the crop and completely degraded to harmless metabolites, 20% is emitted to the air, 2% is emitted to water and 33% is emitted to soil. In Figure 4 the environmental indices calculated on the basis of this alternative estimate of pesticide emissions are compared with those calculated on the basis of the initial estimate of these emissions, i.e. 100% of the pesticides applied to the crops are emitted to water. The calculations were made after lowering the effect scores for aquatic ecotoxicity by a factor 1000 for the initial estimate of pesticide emissions (100% to water), and the effect scores for terrestrial ecotoxicity by a factor 1000 for the alternative estimate of pesticide emissions (the 33% emitted to soil is the main factor in the environmental index in this case).

There is found to be a significant difference in the order of preference of the four margarines between the two options for distributing the pesticide emissions over the various environmental compartments (Figure 4). If it is assumed that 100% of the pesticides applied are

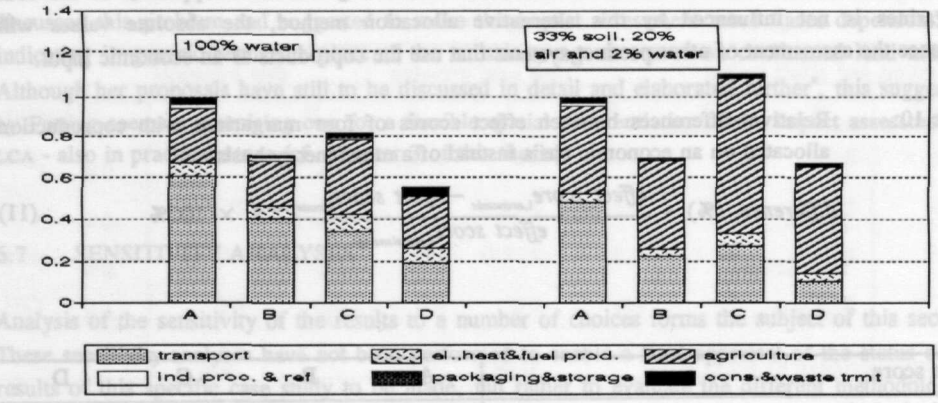

Figure 4: Comparison of the environmental indices of the margarines A, B, C and D based on two different estimates of pesticides emissions (after lowering aquatic ecotoxicity and terrestrial ecotoxicity scores by a factor 1000).

emitted to water, the order of preference is D-B-C-A, while if it is assumed that 33% is emitted to soil, 20% to air and 2% to water the order of preference is D-B-A-C. The difference is due to a combination of reasons: different types of pesticides are assumed to be applied in the life-cycle of each margarine; the ratio between the equivalency factors for terrestrial ecotoxicity for the pesticides considered in this study is different from the ratio between the equivalency factors for aquatic ecotoxicity for the same chemicals; for some pesticides equivalency factors are available for aquatic ecotoxicity but not for terrestrial ecotoxicity and vice versa; and the sustainability factor for terrestrial ecotoxicity according to Eqn. (10) is 4 times higher than that for aquatic ecotoxicity. This combination of causes means that the share of agriculture is higher for the alternative estimate of pesticide emissions than for the initial estimate, although the total amount of pesticides emitted in the alternative estimate is lower, because of the assumption that in that case 45% is degraded. It can be concluded that the manner in which pesticide emissions are distributed over the various environmental compartments significantly influences the results of this particular case study. The distribution should therefore be based as far as possible on proper monitoring.

6.7.3 Equivalency factors for abiotic resources based on reserves and production

In Chapter 5 of this thesis, a new method was described to characterize extractions of resources contributing to depletion of abiotic resources*. In this method it is proposed to measure abiotic depletion by means of physical data on reserves and deaccumulation. Equations have been developed to calculate equivalency factors for these two categories of resources, resulting in so-

* The method described in Chapter 5 is not the last word on this topic of course. For example, at the SETAC-Europe congress in Brussels (April 11-14, 1994) Finnveden presented an interesting and promising method for characterizing extractions of abiotic resources based on the concept of exergy (or entropy).

called *abiotic depletion potentials* (ADP). These ADP-values have been calculated for a wide range of resources. The new aspects of this method compared with the provisional method described in Heijungs *et al.* [6] are:

- extractions of resources are characterized on the basis of both reserves and production rates instead of reserves only;
- data on reserves are based on earth crust concentrations (and suchlike) instead of on economically exploitable reserve estimates;
- in Chapter 5 equivalency factors are presented for all types of resources, while in the provisional method in Heijungs *et al.* [6] equivalency factors are only presented for fossil fuels, uranium and certain heavy metals.

To calculate the effect score for abiotic resource depletion according to the method described in Chapter 5, resources such as rock salt, pyrites, bauxite, phosphate rock and limestone must be translated into their constituent elements using the relative mole weight of the constituent elements. These recalculated extractions of various elements are multiplied by the *abiotic depletion potential* (ADP) of that particular element, values for which are given in Chapter 5, and the results aggregated into the effect score for abiotic depletion. In order to calculate the change in the environmental indices of the margarines according to the method described in Chapter 5, new values for N and D need to be calculated according to this method. The value for N was calculated in Chapter 5 and equals 2.20×10^{12} (kg/yr), which implies that D is half this value, according to the assumption made in Table 3.

Table 11 shows the partial environmental indices due to extraction of abiotic resources. The partial indices are calculated by multiplying the effect scores for abiotic resource depletion by $(N\div D^2)$, according to Eqn. (10) in § 6.5.4. The aggregate value of the environmental indices according to Eqn. (10) amounts to about 1×10^{-7}, and about 1×10^{-9} after lowering the effect scores for aquatic ecotoxicity. Comparing these absolute values with the partial values for abiotic depletion, as stated in Table 11, shows that the environmental indices of margarines A, B, C and D are not influenced by the method chosen, despite the significant change in the absolute values. This is due to the fact that abiotic depletion contributes only marginally to the total environmental indices of the four margarines.

Table 11: Comparison of contributions of abiotic depletion to total environmental index according to Eqn. (10) and effect scores according to two types of characterization methods for abiotic resources (effect scores also given relative to effect score of margarine A)

	A	B	C	D
method of Chapter 5	1.58e-12	1.32e-12	2.00e-12	1.62e-12
relative to margarine A	1.00	0.83	1.27	1.02
method of Heijungs *et al.* [6]	7.62e-14	6.83e-14	1.28e-13	1.09e-13
relative to margarine A	1.00	0.90	1.68	1.43
ratio of Chapter 5 to Heijungs *et al.* [6]	20.7	19.3	15.6	14.9

6.7.4 Uncertainty ranges of equivalency factors

As some uncertainty estimates can be derived from Heijungs *et al.* [6] for GWPs and POCPs, the effect scores for global warming and photochemical oxidant formation have been recalculated using the upper and lower values of these ranges. The results are shown in Table 12.

Table 12: Comparison of effect scores for global warming and photochemical oxidant formation of margarines A, B, C and D for mean, lower and upper values for GWPs and POCPs (effect scores are relative to effect score of margarine A !)

global warming	A	B	C	D
mean value	1.00	0.85	0.88	0.69
lower value	1.00	0.85	0.86	0.67
upper value	1.00	0.86	0.91	0.76
photochemical oxidant formation	A	B	C	D
mean value	1.00	1.00	1.22	1.34
lower value	1.00	0.98	1.25	1.37
upper value	1.00	1.03	1.19	1.30

The differences in results appear to be marginal, but this cannot be generalized as this depends on the number and amount of chemicals emitted during the life-cycle of a given product system that contribute to these impact categories. In this case study the number of such chemicals was very low and the differences were small. The environmental indices of the margarines A, B, C and D, shown in Table 13, change only slightly. The ranking is unaffected.

Table 13: Comparison of environmental indices of margarines A, B, C and D after lowering the aquatic ecotoxicity scores by a factor 1000 for mean, lower and upper values for GWPs and POCPs (indices relative to index of margarine A !)

	A	B	C	D
mean value	1.00	0.72	0.71	0.47
lower value	1.00	0.71	0.70	0.46
upper value	1.00	0.73	0.73	0.51

6.7.5 Alternative equations for calculating the sustainability factor

In this section the influence of two equations used to calculate the sustainability factor is compared. In Section 6.6 the environmental indices of margarines A, B, C and D were calculated according to Eqn. (10). Applying the same data sets, the environmental indices are here calculated

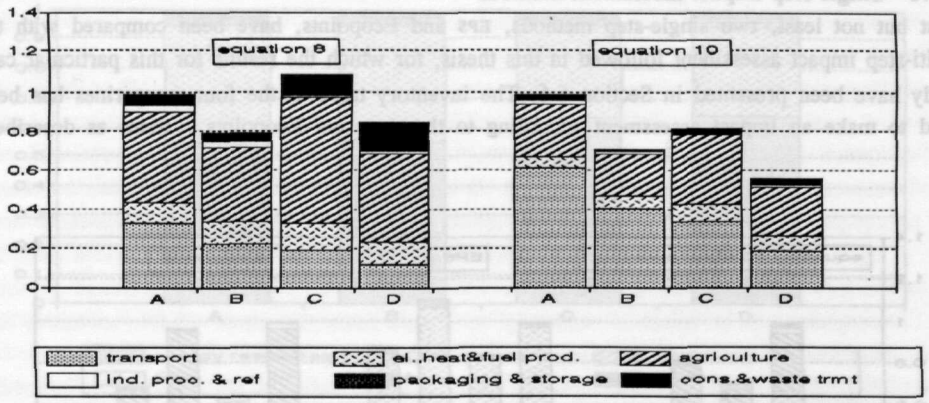

Figure 5: The environmental index of margarine A, B, C and D calculated according to two different equations (after lowering the aquatic ecotoxicity scores by a factor 1000)

according to Eqn. (8) and compared with the indices according to Eqn. (10). The results are shown in Figure 5. The two equations appear to give different results. The sequence of environmental preference according to Eqn. (8) is B-D-A-C and according to Eqn. (10) D-B-C-A. The two different equations cause these differences because they result in different sustainability factors. For example, the change for margarine C can be explained as follows: according to Eqn. (8) the D for nutrification and acidification are (almost) equally highest and as margarine C is the worst on nutrification and the worst but one on acidification, this results in the highest index for margarine C; according to Eqn. (10) the D for acidification is by far the highest and as margarine C is "only" the worst but one on acidification (after margarine A), this results in the highest but one index for margarine C.

A rational choice for either one of these two equations for calculating the sustainability factor is not yet feasible; the choice may well be decisive, although not to a large extent. Moreover, changes in the values for N_i and D_i will probably also have a decisive effect on the results and the correctness of these values needs to be assessed and, if necessary, improved*.

The quantities and units in Eqn. (10) are similar to those in the equation applied by Braunschweig and Müller-Wenk [33] in the "Ecopoints" method. The essential difference between the two equations is that in this paper the equation is applied at the level of the ecoprofile while in the "Ecopoints" method it is applied at the level of the inventory table. The possibility previously suggested [62] - that these two essentially quite different LCA-methodologies may approximate one another - might well become reality.

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* In Heijungs *et al.* [6] a method is proposed for calculating the points at which the outcome reverses owing to a change in the weighting factors applied. In this case, such reversal points could have been calculated for a combined change in the equation and/or the data for N and D, but it would have been almost impossible to determine which of these three possible causes was the main one. For this reason, just one of the potential causes, the type of equation used, has been analyzed here.

6.7.6 Single-step impact assessment methods

Last but not least, two single-step methods, EPS and Ecopoints, have been compared with the multi-step impact assessment followed in this thesis, for which the results for this particular case study have been presented in Section 6.6. The inventory table of the four margarines has been used to make an impact assessment according to the EPS- and Ecopoints systems as described,

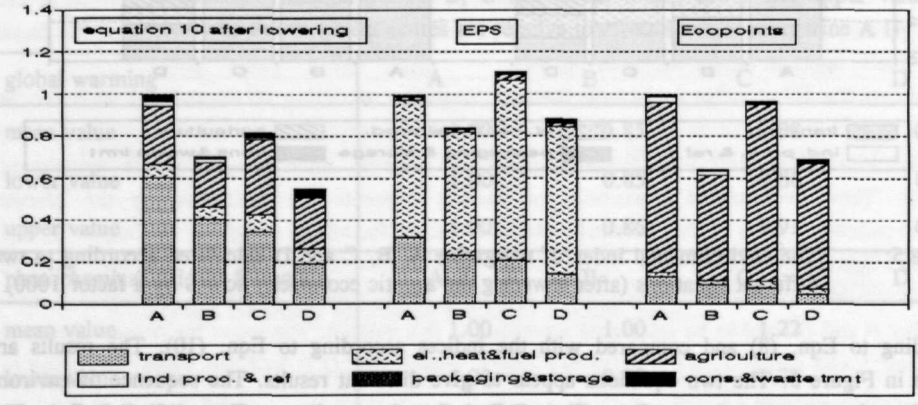

Figure 6: Environmental indices for margarines A, B, C and D according to different impact assessment methodologies (after lowering the aquatic ecotoxicity scores by a factor 1000 for Eqn. (10))

respectively, by Steen and Ryding [32] and by Braunschweig and Müller-Wenk [33]. The results of the environmental indices of margarines A to D relative to margarine A are shown in Figure 6. As neither the EPS nor the Ecopoints approach include the possibility of assessing emissions of pesticides (!), the results of these two methods are unaffected by the erroneously high estimate of pesticide emissions.

As might be expected, it is found that the three impact assessment methods give different results. The sequence of environmental preference according to the environmental theme approach is D-B-C-A (after lowering the aquatic ecotoxicity effect scores by a factor 1000), whereas this sequence is B-D-A-C according to the EPS approach and B-D-C-A according to the Ecopoints approach.

The limited scope of both the EPS and the Ecopoints method is one of the main reasons for the different results. In Figures 7 and 8, stacked bars are shown illustrating the inventory items contributing most to the indices calculated with the EPS and Ecopoints approaches.

It appears that in the EPS method the highest weights are attributed to extractions of fuel resources (oil, gas, coal) and emissions of CO_2, while in the Ecopoints method the highest weights are attributed to emissions of phosphorus. In the multi-step environmental themes approach, the highest weights are attributed to the emissions of pesticides. The three methods thus give completely different results, because of differences in scope and differences in environmental and

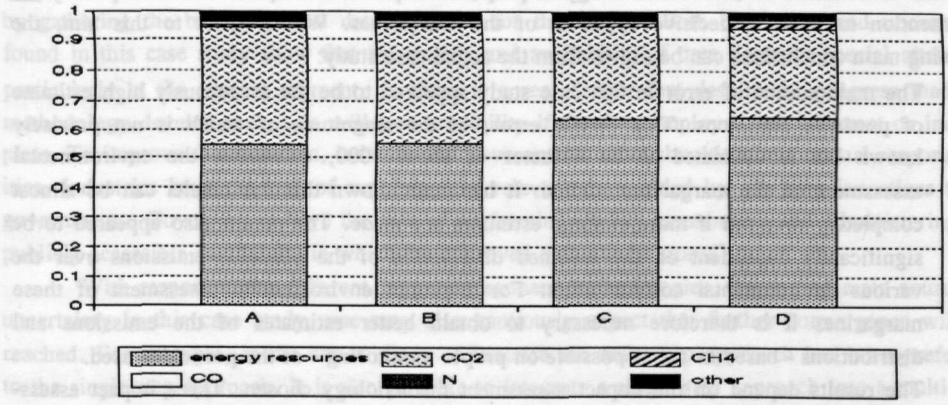

Figure 7: Main inventory items contributing to the environmental indices calculated according to the EPS approach

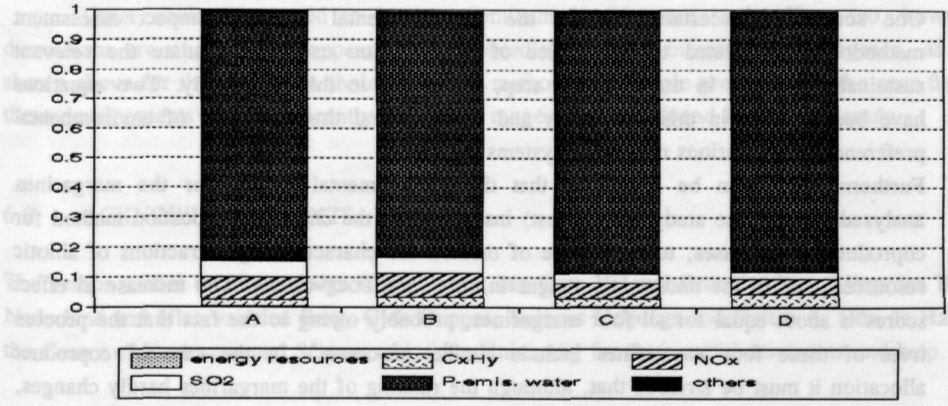

Figure 8: Main inventory items contributing to the environmental indices calculated according to the Ecopoints approach

other considerations. The fact that the differences between the results of the three methods are relatively moderate, is probably pure coincidence.

6.8 CONCLUSIONS

The aim of this methodological study on margarines was to assess to what extent the results of a case study can be influenced by choices in methodology and data in the various methodological

steps and to illustrate the LCA-methodology as proposed in previous chapters. It was explicitly not the intention to make a decisive appraisal of the margarines. With respect to this aim, the following main conclusions can be drawn from the actual case study:

- The main source of error in this case study appeared to be the erroneously high estimate of pesticide emissions. This certain error, of the magnitude of which is not precisely known but is estimated to be a factor of about 1000, dominates the environmental assessment of the margarines studied. It has been shown that the results can be almost completely reversed if more realistic estimates are made. The results also appeared to be significantly dependent on the assumed distribution of the pesticide emissions over the various environmental compartments. For a proper environmental assessment of these margarines it is therefore necessary to obtain better estimates of the emissions and distributions - based as far as possible on proper monitoring - of the pesticides used.
- The results depend on the impact assessment methodology chosen. Three impact assessment approaches have been compared: the multi-step "Environmental Themes" approach, and the single-step EPS and Ecopoints approaches. The results of the three approaches differ significantly, because of the different scopes of the approaches and the different weights (based on environmental and other considerations) given to the various environmental impacts.
- One source of uncertainty within the "Environmental Themes" impact assessment methodology is related to the choice of the equation used to calculate the relevant sustainability factor in the valuation step, as applied in this case study. Two equations have been applied in this case study and it was found that the order of environmental preference of the various margarine systems changed.
- Furthermore, it can be concluded that the environmental indices for the margarines analyzed in this case study are (almost) insensitive to the choice of allocation method for coproduction processes, to the choice of method for characterizing extractions of abiotic resources, and to the uncertainty ranges in GWP- and POCP-values. The increase in effect scores is about equal for all four margarines, probably owing to the fact that the process trees of these four margarines include similar processes*. In the case of coproduct allocation it must be stressed that, although the ranking of the margarines hardly changes, the effect scores of the four margarines may increase from 0.5% to almost 74% if allocation is carried out on an economic instead of a mass basis.

Reading these case study conclusions, the impression might be obtained that the outcomes of an LCA may point in any direction, depending on the particular choices made. Although this impression can never be removed completely as the method can always be abused, the ranges in the results as calculated in this case study can only diminish in the near future. The main "error" in this case study was due to an erroneously high estimate of pesticide emissions. Such large errors can be avoided by improving data quality checks (with the help of normalization, but also by applying "Fingerspitzengefühl") and will be avoided in the future, having learned from previous errors. More regular subjection of data sets to peer review can also certainly help here.

The major uncertainty in this case study is due to the assumed distribution of pesticide

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* For this reason, it is also impossible to generalize the conclusions of this case study.

emissions over the various environmental compartments. This uncertainty can also be diminished by gathering the best empirical data available for this aspect. Both data error and uncertainty found in this case study show that the quality and availability of data deserves special attention, particularly at the present moment, as there is a real danger that despite the progress made in methodology development, LCA will lose credibility if the data employed are incorrect or incomplete. To improve this situation, industry and governments must shoulder their mutual responsibilities. Industries have the task of supplying the relevant data, aided by adequate incentives from governments. Governments have the task of performing quality assessments and administering (a) publicly accessible database(s) in which the approved data can be stored".

With respect to the different impact assessment methodologies, the second main source of uncertainty in this case study, one may also cautiously expect that further convergence will be reached. For example, in the SETAC Code of Practice the multi-step approach has been preferred to the single-step approach. It is also possible to integrate a single-step approach into a multi-step approach. In this case study, and also by Müller-Wenk [63], it was shown, for example, that the basic equation of the Ecopoints method may well be applied in the valuation step of the multi-step approach, by translating inventory data into effect scores by applying the "Environmental Themes" method and effect scores into one overall index by applying the equation of the Ecopoints method.

Other choices appeared to be less dominant in this case study. Here too, however, discussions are ongoing and appear to be proceeding towards increasing convergence, as was shown above for the discussion about allocation methods in multiple processes. In the near future, these developments may significantly reduce the ranges in results found in this case study.

6.9 ACKNOWLEDGEMENTS

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* As will be clear, updates of both inventory and classification data (equivalency factors) to keep abreast of the latest knowledge will always remain necessary.

6.11 Appendix 1 (Inventory items per process contributing > 0% to the acidification score of margarine A)

Margarine A	NO_x	SO_2	NH_3	HCl	acidification
TOTAL	34.48%	64.31%	1.01%	0.20%	100.00%
tractor transport for working on land (T)	1.54%	0.91%	0.00%	0.00%	2.46%
P_2O_5 PRODUCTION (A)	0.02%	0.00%	0.00%	0.00%	0.02%
transport by cargo ship (T)	0.00%	0.01%	0.00%	0.00%	0.01%
transport by lorry, Asia, 15 ton (T)	0.61%	0.06%	0.00%	0.00%	0.67%
H_2O_2 PRODUCTION (I)	0.00%	0.00%	0.00%	0.00%	0.00%
ind. steam production, lignite (E)	0.00%	0.01%	0.00%	0.00%	0.01%
transport by truck (T)	0.09%	0.01%	0.00%	0.00%	0.10%
refining of extracted light oil (E)	0.00%	0.07%	0.00%	0.00%	0.07%
transport by ship (T)	22.33%	55.29%	0.00%	0.00%	77.63%
H_2SO_4 PRODUCTION (I)	0.00%	0.22%	0.00%	0.00%	0.22%
production of bleached sulphate pulp (P)	0.04%	0.10%	0.00%	0.00%	0.14%
pipeline gas transport (T)	0.05%	0.00%	0.00%	0.00%	0.05%
ind. steam production, hard coal (E)	0.02%	0.08%	0.00%	0.00%	0.10%
ind. steam production, e.l. oil (E)	0.11%	0.16%	0.00%	0.00%	0.27%
Boiler house, B&J (E)	0.19%	0.05%	0.00%	0.00%	0.24%
Boiler house, B&J (E)	0.55%	0.16%	0.00%	0.00%	0.71%
prod. of kraft stand.bl./str. paper (P)	0.00%	0.00%	0.00%	0.00%	0.00%
processing of extracted gas (E)	0.00%	0.58%	0.00%	0.00%	0.58%
ind. steam production, gas (E)	0.05%	0.00%	0.00%	0.00%	0.05%
electr. generation, gas (E)	0.04%	0.00%	0.00%	0.00%	0.04%
electr. generation., heavy oil (E)	0.05%	0.26%	0.00%	0.00%	0.31%
electr. generation, lignite (E)	0.06%	0.21%	0.00%	0.00%	0.27%
electr. generation, hard coal (E)	0.13%	0.34%	0.00%	0.00%	0.47%
refining of extracted heavy oil (E)	0.00%	0.17%	0.00%	0.00%	0.17%
production of margarine A (I)	0.00%	0.00%	0.97%	0.00%	0.97%
sea tanker oil transportation (T)	0.03%	0.55%	0.00%	0.00%	0.58%
Rhine ship transportation (T)	0.31%	0.19%	0.00%	0.00%	0.50%
ind. steam production, heavy oil (E)	0.44%	2.38%	0.00%	0.00%	2.82%
transport by lorry, Europe, 15 ton (T)	0.02%	0.01%	0.00%	0.00%	0.03%
diesel production (E)	0.00%	1.45%	0.04%	0.00%	1.49%
transport by lorry, Europe, 6 ton (T)	0.64%	0.28%	0.00%	0.00%	0.92%
transport petrol use, car (T)	4.62%	0.08%	0.00%	0.00%	4.70%
incineration of paper/carton	2.54%	0.68%	0.00%	0.20%	3.43%

Characters after process names refer to life-cycle stage to which the process belongs according to the definition of the author: T=transport; E=electricity, heat & fuel production; A=agriculture; I=industrial processing & refining; P=packaging & storage; C=consumption; W=waste processing

Chapter 7

Concluding discussion and research recommendations

7.1 INTRODUCTION

Work on this thesis started in 1989. At that time there was growing interest in LCA as a tool for assessing and comparing the environmental impact of products considered throughout their entire life-cycle. At the same time, it was clear that the LCA methods of that time were not transparent and that the scientific basis of several parts of the method were in need of improvement. The aim of this thesis was to increase the transparency of LCA as a decision support tool and to improve the scientific basis of LCA, particularly with respect to the methods used for impact assessment.

In terms of the aim of this thesis, the following has been achieved:

- a methodological framework for LCA has been proposed which has largely been adopted by SETAC;
- for the impact assessment component of LCA, a problem-oriented approach ("Environmental Themes Approach") together with methods for calculating a number of specific equivalency factors to characterize extractions and emissions have been proposed, which are now in full discussion within SETAC.

The methodological framework and the impact assessment proposals have been described in Chapter 2, 3, 4 and 5. In Chapter 6 it has been analysed to what extent the results of an LCA can be influenced by assumptions and choices in methodology and data in the various steps of the LCA methodology.

Since 1989 much progress has been made in the international development and harmonization of LCA methodology, but still quite a number of problems and gaps remain. To the extent that these problems relate to choices and assumptions to be made within each LCA case study (e.g. the definition of the functional unit), these problems will never be "solved" but will have to be discussed explicitly in each case study. Other problems, however, can be solved through additional research efforts (e.g. data problems related to the inventory and to the methods for impact assessment).

In this chapter, the state of the art and the unresolved problems of each step of the various components of the methodological framework are discussed, based on the findings of the previous chapters. Finally, some recommendations for further study are made.

7.2 GOAL DEFINITION AND SCOPING

Goal and application

As the goal and application of each LCA study will determine several choices and assumptions to be made during the study, the first step of any case study is to determine its goal and application. The reasons for performing the study have to be clear. This is not only necessary because the goal and application will determine some of the assumptions and choices to be made (see below), but also to ensure clear external communications during and after completion of the study.

Product systems studied

In each case study, the product system(s) under study must be clearly specified, with restrictions unambiguously stated. If, for example, only light bulbs from factory "X" are being studied,

because the factory is paying to do so, this restriction must be stated clearly. If a study concerns a comparison of products, there must be due discussion between the parties involved (researchers, commissioning party, and representatives of government, industry and consumer organizations, etc.) about the alternatives that may usefully be compared. The outcome of this discussion will differ for each study, since it will depend on the views of the parties involved (see the paragraph on Functional Unit for a further discussion of this aspect). The spatial scale and time horizon of the study must also be clearly established, e.g. "light bulbs as available in the Netherlands in 1994".

Scope and depth of the study

An average LCA study will cost a considerable amount of time and money. This amount will depend on the depth of the study, which in turn depends on the intended application of the study. In each study, the level of detail will have to be weighed up against the application.

An LCA can be performed at various levels of detail, in an iterative fashion. One may start with an assessment using superficial or average literature data; this results in a "quick and dirty" assessment. Such an assessment may indicate those points for which a more detailed assessment should be performed, for which the data seem erroneous, or for which a site-specific impact assessment might provide useful additional information, etc. This process of identifying key issues for which a more detailed assessment is needed is referred to as screening. As discussed in the EC debate about the role of LCA in ecolabelling, this process implies the use of one framework and one set of methods and not (or to a much lesser degree) the development of so-called "simplified methods". Screening, as defined here, is related to the intended application of the assessment in two ways. For certain applications only "quick and dirty" assessments are possible in practice. In the beginning of the process of product design, for instance, it is often not possible to perform a detailed assessment, as the product system is not (yet) known in all its details. For other applications, preference for a more or less detailed assessment depends on the possible socio-economic implications of the application. In the case of an ecolabel, for instance, it would seem preferable to undertake a detailed assessment giving as precise and correct as possible results, because of the potential market implications of an ecolabel being granted or refused.

Functional unit

The functional unit describes the main common functional denominator of the product systems studied. Quantification of impacts of the product systems considered takes place on the basis of this functional unit. At the outset of each study, therefore, an explicit discussion about the correct functional unit for the purpose of the study should take place with all the parties involved. Together with the outcomes of the discussion about the product systems to be studied (see above), the definition of the functional unit will determine which product alternatives are to be taken into consideration. The more strictly the functional unit is described, the fewer alternatives will remain. As an illustration, the functional unit "private transport over 100 km" may be specified in greater detail as "private transport over 100 km overland", "private transport over 100 km overland by car", etc., until just one product or service remains. In this way, the discussion about the functional unit is a discussion about including or excluding product alternatives from the study. The results of this discussion definitely influence the possibilities and limitations of the

application of the results of a particular study. The functional unit should therefore be defined in relation to the goal and application of the study and be stated explicitly in the study report to ensure clear external communications during and after completion of the study.

The definition of the functional unit is also user-dependent and context-sensitive. Referring to the case study of Chapter 6, the result of an LCA may be that, from an environmental point of view, a household should prefer small margarine tubs, whereas canteens should prefer large-size margarine wrappers, since not all sizes of packaging will be optimum for all situations.

Furthermore, it was argued in Chapter 6 that the packaging and the product it contains are often related and, in some cases, separation of these two life-cycles may lead to substantial errors. For example, it may be the case that differences in the adhesive properties of the packaging alternatives give rise to different amounts of food spillage.

As the functional unit is based on the premise that a certain function is to be fulfilled, the zero option - non-fulfillment of the function - is often beyond the scope of an LCA study. For the zero option, however, there is no need to perform an LCA and this option can thus always be discussed in relation to the results of a particular case study.

7.3 INVENTORY ANALYSIS

Process tree: boundary between product system and environment system

The definition of the boundary between the product system and the environment system is based on the distinction between economic and environmental processes. The term "economic process" refers to any kind of process producing an economically valuable material, component or product or supplying an economically valuable service such as transport or waste management. This means, for example, that agriculture, forestry and landfill of waste are considered to be economic processes with associated emissions of, for example, all kinds of pesticides. However, the definition of the boundary between the product system and the environment system is not always clear-cut, particularly in studies concerning agricultural products (see Chapter 6). For example, the soil can be considered as part of the economic system with respect to the agricultural production processes using the soil, but it remains part of the environment system with respect to environmental processes, e.g. the accumulation of heavy metals in the soil. If a rational choice of the boundary is not possible, a simple sensitivity analysis can be performed by calculating the results for all options and comparing the differences. The differences can be presented as an uncertainty range of the results.

Process tree: boundary between product system under study and other product systems

A single process is usually related to several economic products, which are often connected to several product systems. Some part of the environmental inputs and outputs of the given process, then, have to be allocated to these different product systems. Three types of such multiple processes have been distinguished: 1) production of co-products; 2) combined waste processing; and 3) open-loop recycling. From the discussion on each of the three types of multiple processes in Chapters 2 and 6, it appears that there is still quite some divergence possible in allocation methods and results. A stepwise procedure to be followed in each of these three allocation cases

(or defined for each type of allocation) giving the different methodological options per step could structure discussions about allocation. A first attempt to establish such a procedure has recently been developed in a SETAC-workshop on allocation.

In addition to this procedure, if more than one method is possible for some or all of the three allocation cases, one should strive for a standard sensitivity analysis, to be applied in all case studies.

Process tree: cut-off of processes

In practice, the infinite regression of processes needed to produce inputs for other processes will have to be cut off somewhere. Well-considered cut-offs are difficult to make, since a fair cut-off can only be made if the environmental impact of the processes being cut off is known. However, cut-offs are often made either because data are not known and it would cost too much time and money to search for them, or because the processes being cut off are expected to have a negligible contribution. One solution to break through this kind of circular reasoning is to learn from the experiences of completed studies. For example, from a number of case studies on light bulbs it is known that the impacts of electricity consumption during use are predominant. This result can be used in "quick and dirty" LCAs of new bulbs and as a rule of thumb for designers, although one should always remain alert to possible other negative environmental impacts of the new bulb system.

Another reason for omitting processes occurs when a difference analysis is being undertaken. In this kind of analysis, two or more product systems are compared and identical parts left out, which may save time and money.

Process data

Acquisition of appropriate and reliable process data constitutes one of the bottlenecks of any LCA. The data used determine the results of the study and thus the sources and quality of the data should be accounted for explicitly in every case study report. Three important aspects relating to data acquisition are: the type of technology which is regarded to be representative for the product studied, the data availability, and the data quality.

The choice of the representative technology depends on the goal, spatial scale and time horizon of the study. The goal of a particular study might be to identify improvement options based on an assessment of the product system(s) as they function at present (state of the art). Then, as suggested in Chapter 2, a reasonable choice would be to take *modern technology* as representative, i.e. the average technology which is currently most commonly installed in the area and the time horizon of the study. For instance, if the glass milk bottle in Europe in 1993 is to be assessed, the average technology commonly installed in Europe in 1993 is considered representative. Note that it is important to date LCAs as technologies change with time.

Based on the assessment of the current state of the product systems studied, improvement options must then be identified. First, the main processes contributing to the total impact of the product system(s) under study will have to be identified. For these processes, an inventory of cleaner technologies (the most modern means, the best practicable means or the best technical means available) may be made and a calculation made of the extent to which the environmental impact of the product system(s) studied can be improved. In this way innovations to product

systems will be encouraged.

The next step is to acquire data for the processes considered representative of the product system(s) studied. Unfortunately, publicly available data sources are still limited. Most data have to be taken from literature sources that may easily be outdated. It would be much better if a public database were established, with datasets being regularly updated. Because of this situation, it will frequently be the case that process data that can be considered representative for the product system(s) studied are unavailable. In such cases it may then be preferable to use the second-best published data, even if they are highly outdated, in order to stimulate the responsible industries to come up with better data.

The last step concerns the quality of the available data. As illustrated in Chapter 6, errors in data may be significant. However, methods for a comprehensive assessment of data quality have not yet been elaborated. Drawing up mass and energy balances per process, "Fingerspitzengefühl" and regular subjection of datasets to peer review can improve this situation. There is an ever-present danger that, despite the progress made in methodology development, LCA will lose credibility if the data used are insufficiently correct or complete. To improve this situation, industry and governments will have to take up their mutual responsibilities. Industries have the task of supplying the relevant data, backed up by adequate incentives on the part of governments. Governments have the task of administering quality assessments and a publicly accessible database in which the approved data can be stored. This database might be built up of separate databases for each economic sector, all using the same format of data storage.

Inventory table

The results of the inventory are presented in the inventory table. Inventory tables can be presented per process, aggregated for groups of processes or aggregated for all processes of the life cycle of the product studied. Non-aggregated results should also be presented since these can be valuable for peer review and for dominance analysis. How to present these large amounts of data in a surveyable form remains a problem, however.

Sometimes (part of the) inventory results are not presented at all, for reasons of confidentiality or because of a fear that the results might be misinterpreted or even abused. One solution to this problem may sometimes be to aggregate the results for e.g. heavy metals or pesticides into one group parameter. However, the results of such LCA studies are not very useful in the public LCA debate, since validation of aggregated results is in fact impossible.

7.4 IMPACT ASSESSMENT

In this thesis a multi-step method for impact assessment has been developed (Chapters 3, 4 and 5) distinguishing classification (or classification and characterization in "Code of Practice" terminology) and valuation. Two methods for performing the impact assessment in only one step (single-step methods) have been compared (see Chapter 6) with the multi-step method developed in this thesis: the EPS approach and the Ecopoints approach. The results of the three approaches can be significantly different, due to the different scopes - the EPS and Ecopoints approaches are elaborated for about 20 different emissions, while the method described in this thesis is elaborated

for more than 500 different types of emissions - and the different weights (based on environmental and other considerations) given to the various environmental impacts.

In the multi-step method a clear distinction is made between environmental science and societal/political preferences in the definition of the separate steps. This is probably also the reason why in the SETAC Code of Practice (COP) the multi-step approach has been preferred. However, this does not mean that single-step methods have been rendered useless. For example, a very practical advantage of single-step methods is that they arrive at one single index for each product alternative studied. This simplifies decision making, whereas most multi-step methods do not yet go this far. As shown in this thesis (Chapter 6), it is also possible to integrate this strong point of single-step methods into the framework of the multi-step framework developed here.

Another well known impact assessment method is the critical volumes approach. In this approach, semi-political standards are applied to aggregate emissions into one score for airborne emissions and one score for waterborne emissions. Besides these scores, a score for energy use and solid waste is often included. In a final valuation, these scores are weighted in relation to one another. This last step has never been formalized. This approach has met with a lot of criticism, because it is based on semi-political standards mixing scientific, economic and technical considerations, and it is a media-oriented approach (air and water), while environmental policy today is more and more multi-media, problem-oriented, offering better potential for a scientifically based impact assessment. Nevertheless, the critical volumes approach has fulfilled an important role in the development of the impact assessment component of LCA. Not in the last place, it has been an important source of inspiration for developing the multi-step, problem-oriented method as described in this thesis.

Classification

The main subject of the classification (according to the "Code of Practice") step of the multi-step impact assessment is the definition of the impact categories. In Chapter 3 of this thesis an extensive list of impact categories that might be included in LCA has been presented. The impact categories refer to the major environmental themes, such as global warming, acidification etc., which explains the fact that this type of impact assessment is often referred to as the "Environmental Themes" approach (impact category = environmental theme = problem type).

On the one hand, the definition of impact categories is a normative task, since they refer to societal values - such as human health, biodiversity and material welfare - which are or may be affected by extractions of resources or emissions of hazardous chemicals. Changes in society's opinion about these values may thus affect the list of impact categories. On the other hand, the precise definition of impact categories is a scientific task, distinguishing exclusive categories based on knowledge about mechanisms and processes in the environment.

Space consumption, energy consumption and final solid waste should not be included as impact categories in their own right in such a list. The environmental problem of space consumption concerns the quality of the space used in terms of, *inter alia*, the degradation of ecosystems and not in terms of the total amount of space used (which is a physical planning problem). It has been proposed in this thesis to classify this aspect under the heading "physical ecosystem degradation". Energy consumption is not an environmental problem as such, but may contribute to a number of impact categories including resource depletion (of both biotic and abiotic energy

resources), global warming, acidification, nutrification and some physical disturbances. The same holds true for final solid waste. Final solid waste is not a problem as such, but rather an economic process ("storage of solid waste") causing emissions to water, air and soil, consuming space and producing methane as a potential energy source. If data on storage of solid wastes are lacking, it seems an appropriate solution to subdivide solid wastes into classes such as chemical wastes, household waste, nuclear wastes, etc, and to aggregate solid wastes into kilograms for each class distinguished.

Characterization

In this thesis it has been proposed to perform characterization according to the environmental themes approach, by multiplying the inventory data (extractions and emissions) by so-called equivalency factors and by aggregating the results of these multiplications into one or more effect score(s) per impact category. Equivalency factors have been proposed and/or developed for a number of impact categories (see Chapters 3, 4 and 5).

The remaining problems concerning this type of characterization relate to the following points:

1 spatial differentiation of equivalency factors;
2 parallel and serial impacts;
3 inadequacy of equivalency factors;
4 lack of data and lack of knowledge;
5 gaps between inventory data and equivalency factors;
6 uncertainties in data.

Spatial differentiation may be relevant if, for example, emissions take place at sea (e.g. by a ship), for which it can be argued that acidification and human toxicity are not as severe problems as on the continent. All attempts to deal with spatial differences in exposure and effects within one impact category will have to deal with the problem that an average LCA may easily include a hundred processes situated all around the world. It will be impossible to make a site-specific assessment for all these processes individually. Spatial differentiation, if possible at all, must therefore be effected in some generic way, for example by dividing the world into, say, ten regions for which region-specific equivalency factors are determined. Examples of spatially differentiated impact assessment are still rare. In this thesis a characterization has been developed without yet introducing any spatial differentiation.

Emitted substances can contribute to more than one impact category, which may be either in parallel or in series to one another. In this thesis three paths have been distinguished:

- *in parallel*, the emission of a substance may potentially contribute to several problems, but actually only contributes to one of these problems;
- *directly in series*, the emission of a substance may have several impacts, one after the other;
- *indirectly in series*, the emission of a substance contributes to another impact through a metabolite of its degradation or through the impact which it primarily causes.

In principle, the distinction between parallel, directly serial and indirectly serial impacts should be considered in the characterization of chemical emissions. The key question is how to handle these different classes of multi-impact emissions. Methods to this end have been proposed in this thesis

but, unfortunately, not all the data needed to apply them are available.

The definition of a number of equivalency factors is still under development and thus not yet adequate. Examples of inadequate equivalency factors are those for nutrification, acidification, ecotoxicity and human toxicity. The factors for these impact categories are incomplete because they do not incorporate an assessment of the fate of the chemicals emitted into the environment. In Chapter 4 a proposal for improving the characterization of ecotoxic and human toxic chemicals by including the fate of the chemicals has been developed. However, its implementation is not yet possible because much of the required data and knowledge is still lacking. For the method described in Chapter 4, data such as vapour pressure, degradation rates per environmental compartment, etc., are needed for each individual chemical. For the most part these data are lacking because knowledge on these aspects is still limited. However, it is to be expected that these data will become available in the near future, since the EC has adopted regulations making producers and importers of chemicals responsible for gathering such data.

The next problem concerns the gaps between inventory data and equivalency factors. Some chemicals are sometimes measured as group parameters in the inventory analysis, for example PAHs and CFCs, while equivalency factors are available for individual chemical species only, for example anthracene and chrysene, or CFC-11 and CFC-12. Other group parameters which are often found in inventories and for which this problem occurs are "organic compounds", "sulphur compounds", "absorbable organic halogens" (AOX) and "hydrocarbons" (C_xH_y). If the individual emission data are not available, the least one can do is to work with the arithmetic means of the individual equivalency factors within that group.

The last problem mentioned is the uncertainty of some equivalency factors. For example, the global warming potential is defined for several time horizons. It is difficult to determine which time horizon is the best for LCA, and probably the best thing to do is to calculate the results of a case study for different time horizons and see what the differences are. Similar uncertainties are encountered with ozone depletion potentials and photochemical oxidant creation potentials. In case studies, sensitivity analyses should be performed to assess the effect of using the upper and lower values of these factors on the results of the case study.

With time, some of these problems can be expected to be solved by new developments in environmental science. Because of this progressive aspect of science, all equivalency factors will have to be updated in due course.

To obtain a better impression of the meaning of the various effect scores, it has been proposed in Chapter 3 to divide the effect scores by the total magnitude of the effect scores for a particular area and given period of time. This step is called "normalization", and the results of this step indicate the extent to which the product system studied contributes to the total magnitude of a given environmental problem. An initial elaboration of this step has been undertaken. The area considered was the whole world and the period considered one year. However, the results of this first elaboration contain quite a number of uncertainties and potential errors due to the fact that a number of data are missing and due to nomenclature problems with respect to chemicals.

Valuation

The valuation component consists of a valuation of the effect scores of the environmental profile and an assessment of the reliability and validity of the results. In principle, social values and preferences dominate in the valuation of the effect scores. In such valuation, it should first be checked whether one alternative is better than or equal to all other alternatives on all scores. If so, further valuation is unnecessary. If such an unweighted comparison does not lead to a result, as will often be the case, and one aims at a conclusive result, a *multicriteria analysis* might be performed. Such an analysis can either be performed on an ad hoc basis by experts or it can be based on a standard list of weighting factors. A quantitative multicriteria analysis based on a standard list is readily applicable and therefore seems to be the most preferable method. However, the main problem here is compilation of such a standard list with a sufficiently broad societal basis.

One way to arrive at such a standard list is to define for each impact category a sustainability factor indicating the distance between the current and the sustainable level of a problem and an inter-effect factor indicating the relative seriousness of the different effects if they actually occur, as proposed in Chapter 6. These two partial weighting factors should be multiplied by one another and by the normalized effect scores. In Chapter 6, a proposal for developing sustainability factors has been elaborated focusing on three questions: a) which parameters are relevant?; b) which data sources are used?, and; c) how are these parameters weighted in relation to one another, or in other words, which mathematical expression is to be applied? It has been argued that the normalized effect scores should be weighted against the current level of the problem, as quantified in the normalization, and against the long-term target standard. Data sources for both these parameters are scarce, however. For the current level this has already been discussed under the heading of Normalization. Data for target values are also hard to obtain, as there is no generally accepted set of target standards available for the different problem types. Provisional data can be taken as a starting point, but the target standards may not yet have a sufficiently broad societal basis. With respect to the mathematical equation, similar problems are encountered. Several equations are possible, as illustrated in Chapter 6. As yet, however, there is no rational basis for choosing any one of the various equations.

The second subject of the valuation component is overall assessment of the reliability (data) and validity (choices and assumptions) of the results of a particular case study. The reliability of the results can be assessed using various techniques. Error analysis yields results with a margin of uncertainty (e.g. 10 ± 2), provided that data (process data, classification factors, etc.) are specified in this form. Process data are usually not provided in this form, but some equivalency factors are. Classical error analysis is thus of only limited significance for LCA, particularly due to the lack of uncertainty margins for process data. Such margins can be estimated in several ways. For a given factory, one might estimate these margins by measuring emissions at different times, say. Another approach would be to take the available data on the same process from different (literature) sources, calculate the average value and determine the maximum deviation as the margin of uncertainty. In the search for the most critical data with respect to reliability, a helpful tool may be the marginal analysis as described in Chapter 3. For the data for which no margins of uncertainty are specified or can be estimated, a marginal analysis can be performed to identify the process data that should be known most accurately because they appear to have a

crucial impact on the results of the particular study.

With respect to assessing the validity of the results, there is the option of performing sensitivity analyses, as illustrated in Chapter 6. The specifications of the products considered, the allocation rules for multiple processes, the different methods for equivalency factors and the various mathematical expressions for calculating sustainability factors are examples of choices and assumptions for which validity assessment might be performed. Such validity assessment should be performed in each study, and consideration might be given to including such a requirement in a subsequent version of the "Code of Practice".

Finally, the reliability and validity of the results can also be enhanced by procedural means, i.e. by an independent peer review of the study. Fortunately, peer reviews are becoming more and more common practice for LCAs.

7.5 IMPROVEMENT ASSESSMENT

Improvement assessment aims to identify possible improvement options. Techniques for this purpose are still under development. Two techniques developed outside this thesis have been briefly discussed here: *dominance analysis* and *marginal analysis*. With these two types of analyses, a number of options can be generated for improving a particular product. To assess the feasibility of these options requires other expertise outside the field of environmental LCA. It is concluded that, with mathematical procedures for identifying improvement options and inclusion of expertise from process technologists and designers, LCA might become an analytical tool for eco-design, supporting continual environmental improvement of products. However, experience with such improvement assessment techniques is still limited. As product improvement is one of the main applications of LCA, such experience should be gained as soon as possible by performing studies applying these and other techniques to identify improvement options.

7.6 RECOMMENDATIONS FOR FURTHER RESEARCH

Based on the concluding discussions in the previous paragraphs, a number of topics for further research can be recommended. These include:

- definition of general guidelines for the required level of detail in relation to the intended application of a study;
- definition of general, relatively simple guidelines for designers, based on experience with comparable LCA studies;
- development of a stepwise allocation procedure for the three types of multiple processes (co-production, combined waste processing and open-loop recycling) under the auspices of e.g. SETAC. From the discussion on each of these three types of multiple processes, it appears that there is still quite some divergence possible in allocation methods and results, although the basic line of reasoning can in principle be quite similar;
- incorporation of mass and energy balancing techniques for quality checks on process data. Guidelines must be developed for this purpose, making it unnecessary, for example, for

mass balances to be made for every single chemical element involved in a process, which seems an extremely time-consuming activity;

- establishment of one or a series of public database(s) containing all published data on economic processes in a common, useful format for LCA studies. Such a database should be regularly updated and could be surrounded by quality assessment procedures, demands on indications of data reliability (e.g. margins of uncertainty), etc.
- assessment of the feasibility and elaboration of proposals to make the impact assessment more area-dependent (or site-specific) by indicating in generic terms the site of the emission (e.g. at sea or on a continent). Area-dependent impact assessment might give more meaningful results, especially for non-global impact categories such as photochemical ozone creation, acidification, human and ecotoxicity, nutrification etc.
- improvement of the equivalency factors for nutrification and acidification by also taking their environmental fate into account (transport, biodegradation, etc.);
- improvement of the quantity and quality of data needed for the normalization step, e.g. by asking the relevant bodies to produce the information needed for this step;
- inventory (brainstorm) and elaboration of possible methods for the valuation of effect scores, including the method based on a sustainability factor and an effect factor as proposed in this thesis;
- performing case studies applying improvement assessment techniques to identify improvement options in order to gain experience with these techniques.

Although this list of research recommendations probably is not an exhaustive one, it might be perceived as already implying a lot of work. However, most of these efforts are one-off jobs, requiring only updates at a later stage.

7.7 EPILOGUE

The aim of this thesis has been to increase the transparency of LCA as a quantitative decision support tool and to improve the scientific basis of LCA, particularly regarding the methods used for impact assessment. With respect to these aims, substantial progress seems to have been made. However, at the same time it has become clear that there are still many gaps and uncertainties in data as well as a number of methodological inadequacies. It might be argued that a method with so many open ends should not yet be applied in practice and that, for the time being, it might be preferable to apply simple qualitative criteria such as types of resources used, recyclability, repairability, etc. Application of such simple qualitative criteria involves implicit choices, assumptions and evaluations, however, without due scope for analysing whether they are correct. The methodology developed in this thesis is an attempt to objectify the discussions about the environmental impact of products by making explicit the choices, assumptions and evaluations involved and by providing opportunities - by playing with data and methods - for identifying the real main sources of a product's environmental impact. This thesis is intended to serve as a contribution to removing some of the deficiencies of quantitative LCA.

Presently, expectations about the role of LCA, and in particular quantitative LCA, in environmental policy are rather high. There is a risk involved here. It should be borne in mind

that LCA is just one of a number of environmental decision support tools available to assess the environmental consequences of anthropogenic activities and help steer them in a more sustainable direction. LCA does not replace these other tools, which include Environmental Impact Assessment (EIA), Risk Assessment (RA), Substance Flow Analysis (SFA), Technology Assessment (TA) and Environmental Audit (EA). LCA provides additional information from a new perspective, complementing that given by these instruments: an assessment of - as far as possible - all environmental impacts related to all processes of the life-cycle of a product. Bearing these limitations in mind, however, LCA can become a very useful tool in the development towards a more sustainable world economy.

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Samenvatting

Dit proefschrift geeft een analyse van de algemene ontwikkelingen op het gebied van de milieugerichte levenscyclusanalyse van produkten (LCA), en doet op basis daarvan enkele methodische voorstellen. Het proefschrift is opgebouwd uit een inleiding (hoofdstuk 1), een beschouwing over de algemene opzet van de LCA (hoofdstuk 2 en 3), een voorstel voor de kwantificering van de effecten van emissies van toxische stoffen (hoofdstuk 4), een voorstel voor de kwantificering van de effecten van het gebruik van grondstoffen (hoofdstuk 5), een gedeelte waarin de stand van de LCA-methodiek bediscussieerd wordt aan de hand van een case-studie over margarines (hoofdstuk 6), en een slotdiskussie waarin de belangrijkste bevindingen van dit proefschrift voor de diverse methodische componenten en stappen nog eens op een rijtje worden gezet en enkele onderzoeksaanbevelingen worden geformuleerd (hoofdstuk 7). Hoofdstukken 2 tot en met 4 bestaan uit gepubliceerde artikelen; hoofdstuk 2 en 3 vormen tezamen één artikel dat in twee delen gepubliceerd is. Hoofdstuk 5 is als artikel ter publicatie aangeboden.

In hoofdstuk 1 wordt een definitie van LCA en een beknopt historisch overzicht van de ontwikkelingen op het gebied van de LCA gegeven.
LCA is een instrument waarbij alle relevante milieu-effecten van de door produktsystemen geleverde functies zo goed mogelijk in kaart worden gebracht. Met de term "produktsysteem" wordt het geheel van processen aangeduid dat nodig is om een produkt zijn functie te kunnen laten vervullen. Met de term "economisch proces" wordt elk proces aangeduid dat een economisch waardevol materiaal, component of produkt produceert of een economisch waardevolle dienst levert zoals bijvoorbeeld transport of afvalverwerking. Aangrijpingspunt van een LCA is een funktie geleverd door een bepaald produktsysteem. Vervolgens worden alle milieu-effecten van alle processen in de levenscyclus van een produkt - van grondstoffenwinning, via produktie van materialen en het produkt zelf, de consumptie van het produkt, tot en met de afvalverwerking van het afgedankte produkt - in beschouwing genomen.
De allereerste LCA's dateren uit het eind van de jaren zestig en het begin van de jaren zeventig. Deze studies concentreerden zich vooral op energie-aspecten. De wortels van de LCA liggen dan ook in de energie-analyses uit die tijd. Deze werden later uitgebreid met analyses van milieuingrepen zoals grondstoffenverbruik, emissies van schadelijke stoffen en produktie van vast afval. Eén van de eerste studies die een dergelijke brede analyse van milieuingrepen omvatte, was een studie naar drankverpakkingen van het Midwest Research Institute, uitgevoerd in opdracht van Coca Cola in de VS. Daarna nam de belangstelling voor LCA enigszins af tot halverwege de tachtiger jaren. De Swiss Federal Laboratories for Materials Testing and Research (EMPA) publiceerde in 1984 een belangrijk rapport. In dit rapport stond een methode beschreven om emissies van schadelijke stoffen voor lucht en water eerst te delen door voor die stoffen geldende beleidsnormen voor respectievelijk lucht en water en vervolgens te aggregeren tot zogenaamde kritieke volumina lucht respectievelijk water. Een vergelijkbare methode is in dezelfde tijd in Nederland door Druijff geïntroduceerd. Bovendien publiceerde de EMPA in hetzelfde rapport een voor die tijd uitgebreide lijst van milieudata met betrekking tot de produktie van een aantal bulkmaterialen. In de negentiger jaren begon pas de werkelijke opleving van de belangstelling voor LCA, hetgeen blijkt uit de aanhoudende reeks publicaties, workshops en congressen.

De belangrijkste coördinerende organisatie in deze hausse aan activiteiten is de *Society of Environmental Toxicology and Chemistry* (SETAC). Zij heeft via haar afdelingen in Europa en de

vs zeer vele workshops georganiseerd, waarbij onderzoekers uit met name Europa en de vs bijeen werden gebracht om allerlei LCA-kwesties te bediscussiëren. Ook de *International Standards Organization* ISO heeft zich op het terrein van de LCA begeven.

Het tot nu toe belangrijkste resultaat van de SETAC-activiteiten was het opstellen van een eerste versie van een "*Code of Practice*". Eén van de belangrijkste elementen van deze "*Code of Practice*" is het methodologische raamwerk. Dit raamwerk bestaat uit vier componenten: *goal definition and scoping*, *inventory analysis*, *impact assessment* (met een nadere onderverdeling in *classification*, *characterization* en *valuation*) en *improvement assessment*.

Het werk voor dit proefschrift is begonnen in 1989, net vóór de hausse aan internationale activiteiten. Dit proefschrift is in interactie met de hierboven genoemde activiteiten, met name die van de SETAC, tot stand gekomen. Enerzijds heeft de SETAC platforms verschaft om de in dit proefschrift vermelde resultaten te presenteren en te bediscussiëren. Anderzijds hebben diezelfde resultaten een belangrijke bijgedrage geleverd aan de discussies in SETAC-verband.

Eén van die bijdragen wordt beschreven in hoofdstuk 2: de definitie van een algemeen methodologisch raamwerk voor LCA's, en een detailbeschrijving van de eerste twee componenten van dit raamwerk.

In 1989, toen het werk aan dit hoofdstuk begon, maar deels ook nu nog, omvatte de term LCA vele verschillende methoden, met elk hun eigen gebreken. Dit leidde dikwijls tot grote spraakverwarring. Het doel van dit methodologische raamwerk was dan de discussies over de verschillende LCA-methoden en hun gebreken te structureren. Het raamwerk bestaat uit vijf componenten: *goal definition*, *inventory*, *classification*, *valuation* en *improvement analysis*.

Het belangrijkste verschil tussen dit raamwerk en het door de SETAC vastgestelde raamwerk (in de "*Code of Practice*") is, dat in het SETAC-raamwerk *classification* en *valuation* geen aparte componenten zijn maar stappen binnen de *impact assessment* component, en dat de *classification* uit dit proefschrift in het SETAC-raamwerk onderverdeeld is in *classification* en *characterization*. Verdere verschillen zijn slechts terminologisch van aard.

De oorzaak van de verschillen tussen beide raamwerken is van historische aard. Het raamwerk uit dit proefschrift werd eerder opgesteld dan het SETAC-raamwerk. In feite heeft het raamwerk uit dit proefschrift als input voor de discussies voor het SETAC-raamwerk gediend, waarbij het enkele kleine veranderingen heeft ondergaan.

In dit proefschrift wordt de onderverdeling van het raamwerk in componenten op sommige punten nog verder uitgesplitst. Zo zijn de meeste componenten onderverdeeld in stappen. Vervolgens worden voor alle componenten en stappen mogelijke methoden bediscussieerd, en soms worden voorstellen ter verbetering gedaan. Sommige voorstellen zijn in detail uitgewerkt, andere zijn alleen in grote lijnen beschreven. In hoofdstuk 2 worden de *goal definition* en de *inventory* verder uitgewerkt.

In de *goal definition* wordt het doel van de studie bepaald. Het doel is afhankelijk van de beoogde toepassing: wordt een produktverbetering nagestreefd, gaat het om een nieuw ontwerp, gaat het om het publiceren van milieu informatie over één of meer produkten, wordt de studie uitgevoerd ten behoeve in het kader van een milieukeuraanvraag, etc. Het type toepassing beïnvloedt de gehele procedure: op welke wijze en hoe vaak de verschillende methodologische componenten doorlopen worden. Bij alle toepassingen is op één of andere manier sprake van een

vergelijking van produkten, waarvoor dus een vergelijkingsbasis vastgesteld moet worden. De functie die wordt geleverd door de te vergelijken produkten speelt daarbij een centrale rol, en om die reden wordt de eenheid van vergelijking vaak "de functionele eenheid" genoemd. Zo is het bijvoorbeeld niet juist 1 pak melk met 1 fles melk te vergelijken, aangezien de eerste slechts 1× wordt gebruikt terwijl de tweede 25× kan worden gebruikt. Een betere vergelijkingsbasis is de verpakking van 1000 liter melk, waarvoor 1000 pakken dan wel 40 flessen nodig zijn. Tenslotte worden in de doelbepaling ook de ruimtelijke schaal (bijvoorbeeld Nederland of Europa) en de tijdsperiode (gaat het om het pak en de fles van nu, van gisteren of van de toekomst) expliciet vermeld.

In de *inventory* is de levenscyclus de leidraad. Om een kwantitatief beeld te krijgen van de milieuingrepen van een produktsysteem, moeten de grenzen tussen het produktsysteem en het (natuurlijke) milieu en de grenzen tussen het produktsysteem en andere produktsystemen worden vastgelegd. Bij het vaststellen van de grenzen tussen het produkt systeem en het milieu doen zich vragen voor als: behoort een produktiebos tot het milieu of is het een economisch proces binnen het produktsysteem waarvoor meststoffen, bestrijdingsmiddelen en ruimte nodig zijn; behoort afvalverwerking tot het milieu of is het een economisch proces dat ruimtebeslag kost, dat emissies naar lucht, water en bodem veroorzaakt, en misschien wel methaan produceert dat weer als energiebron gebruikt kan worden? In dit proefschrift wordt betoogd dat bosbouw en afvalverwerking economische processen zijn met hun eigen milieuingrepen en dus als zodanig behandeld dienen te worden in een LCA.

Bij het vaststellen van de grenzen tussen het bestudeerde produktsysteem en andere produktsystemen, concentreren de vragen zich rondom drie verschillende situaties: vervaardiging van co-produkten, gecombineerde afvalverwerking, en open-loop recycling. In alle drie de gevallen leveren de processen van het bestudeerde produktsysteem niet alleen een economische output aan dat systeem, maar ook tenminste één economische output aan een ander produktsysteem. Dan moeten de milieuingrepen van dat specifieke proces dus op één of andere manier toegerekend worden aan de verschillende produktsystemen. Toerekening naar rato van een fysische eenheid zoals bijvoorbeeld de massa van de betrokken economische outputs, is een envoudige manier, maar de resultaten van een dergelijke toerekening zijn sterk afhankelijk van de gekozen eenheid (bijvoorbeeld: kg of mol). Toerekening naar rato van de economische waarde geeft weliswaar de waarde die de maatschappelij toekent aan de betrokken economische outputs goed weer, maar vereist extra gegevens over de prijzen van de outputs.

Aan elk economisch proces liggen diverse andere economische processen ten grondslag. In de *inventory* moet worden bepaald hoe de in principe oneindige regressie van processen (waarbij de één weer nodig is voor de ander) binnen een produktsysteem wordt afgekapt. Hier lijken nauwelijks gouden regels voor te ontdekken.

Ook het volgende punt vereist een beredeneerde keuze. Voor elk proces is er een keuze uit meer alternatieven omdat er over het algemeen meer dan één technologie beschikbaar is om hetzelfde produkt te maken. Met betrekking tot de keuze tussen deze processen, en dus tussen de procesdata, wordt voorgesteld om uit te gaan van de recent geïnstalleerde, gemiddelde technieken. Daarnaast hangt deze keuze af van de schaal van de vergelijking (gaat het om één bedrijf, Nederland, Europa, etc.).

Het resultaat van de inventarisatie heet de inventarisatietabel: een lijst van milieuingrepen,

zoals onttrekkingen van grondstoffen en emissies van schadelijke stoffen naar lucht, water en bodem.

Bij een vergelijking van een aantal afgeronde studies naar melkverpakkingen bleken deze in vele aspecten te verschillen. Zo was de functionele eenheid, en met name de specificatie daarvan, in alle studies verschillend, verschilden de definities van in principe dezelfde produktsystemen, verschilden de keuzen van de processen en data, etc. De resultaten van de verschillende studies liepen als gevolg hiervan sterk uiteen.

In hoofdstuk 3 worden de overige drie componenten van het eerder genoemde methodische raamwerk in detail besproken: *classification*, *valuation* en *improvement analysis*.

In de *classification* wordt de bijdrage van de in de *inventory* berekende milieuingrepen, aan een aantal algemeen erkende milieuproblemen (=milieuthema's) gekwantificeerd. Het resultaat wordt wel het milieuprofiel van een produkt genoemd, en bestaat uit een aantal zogeheten effectscores. In de ontwikkeling van de classificatie kunnen vier stappen worden onderscheiden: 1. de definitie van algemeen erkende milieuproblemen die in een LCA beschouwd zouden moeten worden; 2. de definitie van classificatiefactoren (hierna equivalentiefactoren genoemd in analogie met SETAC terminologie) die de bijdragen van één gekwantificeerde eenheid van een milieuingreep aan een bepaald milieuprobleem weergeven; 3. de vermenigvuldiging van de ingrepen met hun equivalentiefactoren en de aggregatie van de resultaten van deze vermenigvuldiging in een aantal effectscores; en 4. de "normalisatie" van de effectscores.

Achttien milieuproblemen zijn onderscheiden, die zijn onder te verdelen in drie categorieën: uitputting, vervuiling en verstoring. Of al deze probleemtypen in een bepaalde case-studie aan de orde zullen komen, hangt af van de beschikbaarheid van inventarisatiedata en equivalentiefactoren. Daarnaast kan een effectscore natuurlijk ook nul zijn als het produktsysteem eenvoudigweg niet bijdraagt aan dat probleem.

In dit proefschrift zijn voorstellen gedaan voor equivalentiefactoren die tot voor kort nog geheel ontbraken. Zo zijn voor vervuilingsproblemen zoals het broeikaseffect, de afbraak van de ozonlaag, smogvorming, verzuring, humane toxiciteit, ecotoxiciteit en vermesting equivalentiefactoren opgesteld voor zoveel mogelijk verschillende stoffen. Daarbij kon voor een aantal problemen zoals het broeikaseffect, de afbraak van de ozonlaag, smogvorming en verzuring volstaan worden met *thorough scientific shopping*. Dit winkelen leverde respectievelijk de *Global Warming Potential* (GWP), de *Ozone Depletion Potential* (ODP), de *Photochemical Oxidant Creation Potential* (POCP) en de *Acidification Potential* (AP) als equivalentiefactoren op. Voor andere probleemtypen ontbraken pasklare methoden, zodat daarvoor methoden moesten worden ontworpen. Ook voor uitputtings- en verstoringsproblemen bleken pasklare equivalentiefactoren schaars. In dit proefschrift zijn voorstellen voor enkele van deze ontbrekende equivalentiefactoren gedaan. Zo wordt in hoofdstuk 4 een voorstel voor de kwantificering van de effecten van emissies van toxische stoffen, en in hoofdstuk 5 een voorstel voor de kwantificering van de effecten van het gebruik van grondstoffen beschreven. Daarmee ligt het zwaartepunt van dit proefschrift ook in de classificatie component.

Ondanks deze voorstellen blijft meer onderzoek hieraan, en ook verbetering van een aantal reeds ontwikkelde equivalentiefactoren, nodig, en wel bij voorkeur in daarvoor opgerichte (bijvoorbeeld, voor het broeikaseffect, het Intergovernmental Panel for Climate Change, de IPCC)

of nog op te richten gespecialiseerde wetenschappelijke discussiefora.

Om een betere indruk te krijgen van de betekenis van de verschillende effectscores, kunnen ze worden gedeeld door de totale omvang van de betreffende problemen uitgedrukt in dezelfde scores. Deze stap wordt normalisatie genoemd en hiermee kan een beeld worden verkregen van de mate waarin het onderzochte produktsysteem bijdraagt aan de totale omvang van de milieuproblemen. Ook kunnen op deze manier verschillen op dit punt tussen de diverse effectscores worden achterhaald.

De *valuation* component bestaat uit een evaluatie van de effectscores van het milieuprofiel en een beoordeling van de betrouwbaarheid en validiteit van de onderzoeksresultaten. Maatschappelijke waarden en preferenties spelen een belangrijke rol in de evaluatie van de effectscores. Eerst wordt nagegaan of één produktalternatief op alle criteria beter of gelijk scoort in vergelijking met de andere alternatieven. Als dat het geval is, is verdere weging niet noodzakelijk. Als een dergelijke ongewogen evaluatie niet tot een duidelijk resultaat leidt, wat vaak het geval zal zijn, zal een weging van de effectscores nodig zijn - tenminste, als men een concluderend resultaat wil. Of dit laatste het geval is, is afhankelijk van het doel en de toepassing van de studie. Bij deze weging kan een kwalitatieve of kwantitatieve multi-criteria analyse worden uitgevoerd, waarbij gebruik kan worden gemaakt van meningen van experts of een standaardlijst van weegfactoren. De kwantitatieve multi-criteria analyse op basis van een standaardlijst van weegfactoren die bijvoorbeeld jaarlijks wordt bijgesteld, is snel, herhaalbaar en eenvoudig toepasbaar. Dit zijn belangrijke voordelen bij de milieubeoordeling van produkten, aangezien reeds miljoenen verschillende produkten op de markt zijn en er dagelijks vele bijkomen. Een dergelijke geformaliseerde weging lijkt voor LCA dan ook de voorkeur te hebben. Het grootste probleem is daarbij het verkrijgen van een maatschappelijk breed gedragen standaardlijst van weegfactoren.

Een evaluatie van de milieuprofielen zonder een beoordeling van de betrouwbaarheid en de validiteit van de resultaten, is weinig zinvol. Methoden hiervoor ontbreken nog grotendeels, wat mede veroorzaakt wordt door het ontbreken van betrouwbaarheidsintervallen in de meeste databronnen. Door Heijungs is voorgesteld om in die gevallen een marginale analyse toe te passen, waarmee die data kunnen worden opgespoord die de grootste invloed hebben op het eindresultaat, en waarvoor dus de grootste nauwkeurigheid zou moeten worden nagestreefd. Er zal zeker meer aandacht aan dit aspect besteed moeten worden, om de geloofwaardigheid van LCA-studies te vergroten.

In de *improvement analysis* worden opties ter verbetering opgespoord. Daartoe zijn tot op heden twee complementaire technieken beschikbaar: de dominantieanalyse en de marginale analyse. Met deze twee technieken kunnen een aantal aangrijpingspunten voor de verbetering van het onderzochte produktsysteem in kaart worden gebracht. Vervolgens zal technologische, financieel-economische en andere kennis, die buiten het veld ligt van de milieugerichte LCA, moeten worden toegevoegd om te beoordelen welke verbeteringen wel en welke niet haalbaar zijn. Met behulp van de genoemde technieken en de aanvullende kennis van buiten het veld van de milieugerichte LCA, kan LCA een analytisch instrument worden voor milieugericht ontwerpen en op die manier een continue verbetering van produkten stimuleren.

Zoals reeds eerder aangekondigd, wordt in hoofdstuk 4 een nieuwe methode voorgesteld voor de definitie van equivalentiefactoren voor voor humane toxiciteit en ecotoxiciteit.

De definitie van goede equivalentiefactoren voor humane toxiciteit en ecotoxiciteit is één van de moeilijkste vraagstukken in de classificatie. De eerder genoemde kritieke volumina-benadering is tot nu toe de praktijk geweest. Zoals reeds uitgelegd, worden in deze benadering emissies van potentieel toxische stoffen naar lucht en water opgeteld tot zogenaamde kritieke volumina lucht respectievelijk water. Daarbij worden blootstellingsroutes en milieuprocessen buiten beschouwing gelaten; de effectbeoordeling is gebaseerd op semi-politieke normen voor stoffen. Deze benadering zou kunnen worden verbeterd door langs twee wegen equivalentiefactoren voor humane toxiciteit en ecotoxiciteit te ontwikkelen: een blootstellingsdeel waarin emissies van stoffen worden vertaald in de concentratie waaraan een receptor kan worden blootgesteld, en een effectdeel waarin concentraties aan effecten worden gerelateerd via de drempelwaarde (drempelconcentratie) waaronder nog net geen nadelige effecten bij de receptor optreden. Beide delen zullen per stof bepaald moeten worden en zoveel mogelijk op wetenschappelijke modellen en empirische data gebaseerd moeten zijn. Dit lijkt ook een realiseerbare doelstelling.

Zo zou het blootstellingsdeel voor zowel humane toxiciteit als ecotoxiciteit berekend kunnen worden met behulp van de zogeheten multimedia-milieumodellen ontwikkeld door Mackay. Deze multimedia-modellen kunnen echter niet direct worden toegepast op de emissies zoals deze in de inventarisatie zijn gekwantificeerd, omdat deze emissies niet zijn gespecificeerd naar de dimensie tijd. De in de inventarisatie berekende emissies van een stof vinden plaats met een onbekende snelheid. In een LCA worden de emissies gerelateerd aan een functie, en dat heeft als gevolg dat de tijdsdimensie eruit wordt gedeeld. LCA-emissies zijn dus pulsen (in kg). In multimedia-modellen zijn tijdsafhankelijke processen zoals afbraak en transport opgenomen; deze modellen werken met emissies als fluxen (in kg/dag bijvoorbeeld). Een mogelijke oplossing voor dit flux-puls probleem is het kiezen van een referentiestof en vervolgens het berekenen van een dimensieloze equivalentiefactor per stof, analoog aan de ODP-, GWP- en POCP-concepten.

Bij de uitwerking van het effectdeel voor humane toxiciteit, moet rekening gehouden worden met de vele mechanismen en de vele verschillende typen effecten, en tevens met carcinogene en mutagene stoffen waarvoor geen drempelwaarde kan worden afgeleid. In dit proefschrift wordt voorgesteld deze problemen op te lossen door de laagste drempelwaarde - oftewel het eerst optredende effect- in beschouwing te nemen, en voor carcinogene en mutagene stoffen virtuele drempelwaarden af te leiden door "toelaatbare" verhoogde risico's te definiëren. Op deze manier kan per stof een zogenaamde HTP (*human toxicity potential*) worden berekend. Zo'n HTP geeft de humane toxiciteit weer van een emissie van een bepaalde stof ten opzichte van de humane toxiciteit van een gelijke emissie van een referentiestof.

Bij de uitwerking van het effectdeel voor ecotoxiciteit, moeten dezelfde problemen worden opgelost als bij humane toxiciteit. Bovendien moet bij een beoordeling van ecotoxische effecten op een of andere manier rekening worden gehouden met de veelheid aan soorten binnen ecosystemen. Een oplossing van deze problemen kan op dezelfde wijze worden geformuleerd als voor humane toxiciteit, zij het dat voorts ook drempelwaarden voor ecosystemen moeten worden afgeleid van toxiciteitsdata over individuele soorten. Hiervoor zijn verschillende methoden beschikbaar en in ontwikkeling. Verder wordt in dit hoofdstuk voorgesteld om een onderscheid te maken tussen terrestrische en aquatische ecosystemen, vanwege de verschillende in deze media voorkomende soorten en vanwege de verschillende routes langs welke deze soorten aan emissies van schadelijke stoffen worden blootgesteld. Omdat specifieke toxiciteitsdata voor ecosystemen in de waterbodem

en voor blootstelling via de lucht ontbreken, worden deze twee compartimenten voorlopig nog buiten beschouwing gelaten. Volgens de hier voorgestelde methode kunnen voor iedere stof een zogenaamde TETP (*terrestrial ecotoxicity potential*) en een AETP (*aquatic ecotoxicity potential*) worden berekend. De TETP en de AETP geven de terrestrische resp. aquatische toxiciteit weer van een emissie van een bepaalde stof ten opzichte van de terrestrische resp. aquatische toxiciteit van een gelijke emissie van een referentiestof.

Met nadruk wordt erop gewezen dat de HTP-, TETP- en AETP-benaderingen zich nog in een vroeg stadium van ontwikkeling bevinden en dat nog voor geen enkele stof concrete waarden zijn afgeleid. Eén van de problemen die nog overwonnen moeten worden, is de beperkte toepasbaarheid van de multimedia-modellen. Deze modellen zijn momenteel alleen toepasbaar op de meeste organische stoffen, maar bijvoorbeeld nog niet op metalen, metaalverbindingen, zouten en polymeren. Op dit punt bestaan er wel aanzetten tot oplossingen, maar er zal nog wel enig onderzoekswerk verricht moeten worden alvorens die oplossingen ook in de praktijk toepasbaar zullen zijn. Een ander belangrijk probleem is de beperkte beschikbaarheid van voor multimedia-modellen benodigde data per stof. In principe is dit dataprobleem oplosbaar, maar de voortgang op dit gebied verloopt zeer traag.

In hoofdstuk 5 van dit proefschrift wordt, zoals gezegd, een voorstel uitgewerkt voor de definitie van equivalentiefactoren met betrekking tot uitputting van grondstoffen. In dit hoofdstuk wordt voorgesteld uitputting van grondstoffen onder te verdelen in uitputting van abiotische en biotische grondstoffen. De equivalentiefactoren voor grondstoffen die aan één van deze problemen bijdragen, worden berekend op basis van fysische data over de reserves en de desaccumulatie van die grondstoffen. Desaccumulatie is gedefinieerd als de jaarlijkse produktie (onttrekking) minus de jaarlijkse regeneratie van een grondstof. In het hoofdstuk worden een aantal vergelijkingen uitgewerkt waarmee de equivalentiefactoren kunnen worden berekend op basis van deze data. Dit heeft geleid tot de definitie van zogenaamde *abiotic depletion potentials* (ADP) en *biotic depletion potentials* (BDP). Vervolgens worden voor bijna alle elementen uit het periodiek systeem ADP's berekend. ADP's voor verbindingen kunnen worden afgeleid van de ADP's van de samenstellende elementen, door de verhouding van deze elementen in die verbinding te bepalen en deze verhouding met de ADP's van die elementen te vermenigvuldigen.

De methode voor de classificatie van grondstoffen die in dit proefschrift wordt gepresenteerd, is slechts één van de vele mogelijke methoden. Daarbij is het interessant te signaleren dat er opvallende convergentie is tussen de in dit proefschrift uitgewerkte methode en de methoden die werden voorgesteld op een de LCA-workshop van 1-7 februari 1992 in Sandestin (VS). De aard van de uitputtingsproblematiek maakt het feitelijk onmogelijk "de correcte" methode vast te stellen, aangezien die correctheid in het geval van uitputting niet empirisch getoetst kan worden. Dit geldt zowel voor de definitie van met name het abiotische uitputtingsprobleem als voor de keuze van de mathematische vergelijking. Voor het abiotische uitputtingsprobleem zijn er tenminste twee definities mogelijk. Het abiotische uitputtingsprobleem kan gedefiniëerd worden als een afname van de beschikbaarheid van primaire natuurlijke grondstoffen, of als een moeilijkere beschikbaarheid van grondstoffen in het algemeen. In het laatste geval tellen grondstoffen die met veel of weinig moeite uit bijvoorbeeld afval herwonnen kunnen worden, ook mee als reserve. In dit proefschrift is met de eerstgenoemde definitie gewerkt. Wanneer met de tweede definitie gewerkt

zou worden, zou het zinvol zijn entropie aan de rij benodigde data toe te voegen. De keuze van de "correcte" mathematische vergelijking voor de berekening van de classifiactiefactoren is eveneens lastig, zo niet onmogelijk. Tot welke macht de verschillende parameters in deze vergelijking worden verheven, is tot op zekere hoogte een willekeurige zaak.

Tenslotte kan worden opgemerkt dat de meest gewenste dataset voor reserves van grondstoffen - namelijk, de uiteindelijk technisch (niet economisch) te winnen hoeveelheid uit aardkorst, zee en atmosfeer - per definitie nooit beschikbaar zal komen, omdat die afhangt van toekomstige technische ontwikkelingen. Er zal dus altijd met een *second best* dataset gewerkt moeten worden.

In hoofdstuk 6 van dit proefschrift is een methodologische case-studie uitgevoerd. Deze case-studie is uitgevoerd om een indruk te verkrijgen in welke mate de resultaten ervan worden beïnvloed door de gemaakte keuzen in methodologie en data, in de verschillende componenten en stappen. De case-studie dient tevens als een illustratie van de diverse methodologische voorstellen. Vier verschillende margarines, die gebruikt kunnen worden voor bakken, braden en het smeren van brood, zijn beoordeeld en vergeleken. Het was niet de bedoeling tot een definitief eindoordeel te komen; de studie was puur als methodologische exercitie bedoeld. Om de invloed van methodologische en dataset-keuzen op het eindresultaat te bepalen, zijn gevoeligheidsanalyses uitgevoerd met betrekking tot de volgende aspecten: schattingen van pesticidenemissies, *impact assessment*-methoden, valuatiemethoden, methoden voor de toerekening van coproduktie, methoden voor de classificatie van grondstoffen, en datasets voor *Global Warming Potentials* en *Photochemical Ozone Creation Potentials*.

Eerst zijn resultaten berekend op basis van een aantal default opties voor de genoemde keuzepunten. Daarbij is een grote fout in de gegevens over de pesticidenemissies aan het licht gekomen. De precieze omvang van deze fout is niet bekend, maar deze wordt geschat op een factor 1000. Deze fout domineert de resultaten volledig; als een meer realistische schatting van de pesticidenemissies wordt gebruikt, ontstaat een tegenovergesteld beeld.

Bij de gevoeligheidsanalyses bleken de resultaten in belangrijke mate afhankelijk van de veronderstelde verdeling van de pesticidenemissies over de diverse milieucompartimenten (bodem, water, lucht). Zowel de schatting van de pesticidenemissies als de veronderstelde verdeling van deze emissies zullen dus nadere aandacht moeten krijgen, alvorens een zinvolle eindbeoordeling van de verschillende margarines mogelijk is.

De resultaten bleken voorts afhankelijk te zijn van de gekozen *impact assessment*-methode. Drie methoden zijn vergeleken: de meerstappen-methode zoals die in dit proefschrift is beschreven (ook wel aangeduid met de term "milieuthematische" benadering); de éénstaps EPS-methode, een in Zweden ontwikkelde methode; en de éénstaps Ecopunten-benadering, een in Zwitserland ontwikkelde methode. De verschillen tussen de resultaten van deze drie methoden zijn significant, wat niet zo'n verrassing is omdat de verscheidenheid van in beschouwing genomen emissies sterk uiteen loopt (zo kunnen met de EPS- en de Ecopunten-methoden pesticidenemissies niet worden beoordeeld), en de gewichten die aan die emissies worden toegekend (op basis van milieuoverwegingen en ook andere overwegingen) ook sterk verschillen.

Binnen de *impact assessment* methode die in dit proefschrift wordt beschreven (de "milieuthematische" benadering) is er nog een andere bron van onzekerheid, die verband houdt

met de keuze van de vergelijking van de zogenaamde duurzaamheidsfactor in de valuatie. De duurzaamheidsfactor geeft aan hoever het huidige niveau van milieu-effecten verwijderd is van een duurzaam niveau. In deze methodologische case-studie is één methode uitgewerkt om te komen tot een standaardlijst van dergelijke duurzaamheidsfactoren. De vergelijking waarmee huidige en duurzame niveaus ten opzicht van elkaar worden gewogen, is daarbij, gegeven een aantal randvoorwaarde, een open keuze. De resultaten van twee vergelijkingen zijn in deze case-studie vergeleken: de uitkomsten liepen slechts weinig uiteen.

De keuzen die gemaakt kunnen worden met betrekking tot methoden voor de toerekening van coproduktie, methoden voor de classificatie van grondstoffen, en datasets voor *Global Warming Potentials* (GWP)en *Photochemical Ozone Creation Potentials* (POCP), bleken nauwelijks van invloed op de eindresultaten. Wat betreft de methoden voor de classificatie van grondstoffen en de datasets voor GWP's en POCP's betreft, verschilden de eindresultaten in absolute zin slechts marginaal. Bij de toerekeningsmethoden voor coproduktie ligt de zaak anders. De toegepaste gevoeligheidsanalyse gaf aan dat de effectscores van de verschillende margarines voor alle milieuproblemen aanzienlijk kunnen verschillen. De veranderingen zijn echter voor alle vier de margarines ongeveer even groot, wat te maken heeft met het feit, dat de procesbomen van de vier margarines zeer veel op elkaar lijken. Het gevolg is dat ook de toerekeningsmethode voor coproduktie nauwelijks invloed heeft op de eindresultaten.

Op basis van deze uitkomsten zou men de indruk kunnen krijgen dat de resultaten van een LCA volkomen afhankelijk zijn van de gemaakte keuzen. Alhoewel deze indruk nooit helemaal kan worden weggenomen, omdat methoden ook misbruikt kunnen worden, zullen toch naar verwachting de verschillen in resultaten in de toekomst kleiner worden. De grootste marge in de resultaten van deze studie is te wijten aan de kwaliteit van de pesticidendata. Deze fouten kunnen worden voorkomen door het verbeteren van de datakwaliteitsbeoordelingen en door *peer reviews*. In het algemeen zal er meer aandacht moeten worden besteed aan de kwaliteit en de verkrijgbaarheid van de voor LCA benodigde data. Dit geldt met name voor de inventarisatie, maar zeker ook voor de classificatie en de valuatie. Op dit moment bestaat er het reële gevaar dat LCA (minder) ongeloofwaardig zal worden omdat de gebruikte data niet correct, of op belangrijke punten onvolledig zijn.

Tenslotte worden in hoofdstuk 7 de belangrijkste bevindingen van dit proefschrift bediscussieerd per component en stap van het methodische raamwerk. Sinds 1989 is er belangrijke vooruitgang geboekt op het gebied van de ontwikkeling van de LCA-methodologie. De voornaamste bijdragen die dit proefschrift daaraan heeft geleverd zijn het voorstel voor het methodologische raamwerk, de probleemgerichte (themagerichte) opzet van de classificatie, en de voorstellen voor een aantal specifieke equivalentiefactoren. Het voorgestelde methodologische raamwerk is in SETAC-verband bediscussieerd en in aangepaste vorm overgenomen (zie boven). De probleemgerichte opzet van de classificatie en de voorstellen voor equivalentiefactoren zijn nog onderwerp van internationale wetenschappelijke discussie.

Echter, niet alle methodische problemen zijn opgelost. Voor zover deze problemen te maken hebben met keuzen en aannamen die gedurende een LCA-studie gemaakt moeten worden, zullen deze problemen ook nooit "opgelost" kunnen worden maar zal in iedere studie opnieuw verantwoord moeten worden hoe er mee omgegaan is. Dit geldt bijvoorbeeld in sterke mate voor het doel van de studie en in het bijzonder voor de keuze van de funktionele eenheid, maar ook

voor de keuze van de systeemgrenzen, de representatieve technologie, de beschouwde milieuproblemen en de methoden om de bijdragen van de bestudeerde produktsystemen aan deze problemen te kwantificeren en te wegen. Andere overgebleven methodische problemen zouden opgelost kunnen worden door verder onderzoek uit te voeren. Aan het slot van dit proefschrift worden hiervoor een aantal suggesties gedaan met betrekking tot versnelling van de methodiek, verbetering van onderdelen van de methodiek en verbetering van de kwaliteit en beschikbaarheid van data.

Summary

This doctoral thesis analyses general developments in the field of environmentally oriented product life-cycle assessment (LCA) and goes on to present several methodological proposals. The thesis comprises an introduction (Chapter 1), a discussion of the general design of LCA (Chapters 2 and 3), a proposal for quantifying the impacts of emissions of toxic substances (Chapter 4), a proposal for quantifying the impacts of resource extraction (Chapter 5), a section in which the status of the LCA method is discussed with reference to a case study on margarines (Chapter 6) and a concluding discussion in which the main findings of this thesis concerning the various methodological components and steps are recapitulated and several recommendations made for further study (Chapter 7). Chapters 2 to 4 consist of published articles; Chapters 2 and 3 constitute a single article published in two parts. Chapter 5 has been offered for publication.

Chapter 1 provides a definition of LCA and a brief historical review of developments in the field of LCA.

LCA is an instrument with which to assess, as well as possible, all relevant environmental impacts of the functions provided by product systems. The term "product system" is understood to mean the entire chain of processes required for a product to fulfil its function. The term "economic process" refers to any kind of process producing an economically valuable material, component or product or an economically valuable service such as transport or waste management. LCA takes as its starting point the function provided by a given product system and goes on to consider all the environmental impacts of all the processes involved in the life-cycle of the product (system), from resource extraction, through production and processing of materials and the product itself and product consumption to waste processing of the disposed product.

The first LCA studies date back to the late sixties and early seventies. These studies focused mainly on energy aspects, and the roots of LCA are thus to be found in the energy analyses of those years. These analyses were later extended to encompass environmental interventions such as resource consumption, toxic emissions and solid waste arisings. One of the first studies to comprise such a broad analysis of environmental interventions was a study of beverage containers conducted by Midwest Research Institute for the Coca Cola Company in the USA. Subsequently, interest in LCA waned somewhat until the mid-eighties. In 1984 the Swiss Federal Laboratories for Materials Testing and Research (EMPA) published an important report. It described a method for dividing airborne and waterborne toxic emissions by political standards for air and water, respectively, and aggregating them into so-called "critical volumes" of air and water. In the Netherlands a similar approach was simultaneously and independently developed by Druijff. In the same report EMPA also presented a list of environmental data on the production of a number of bulk materials that was quite extensive for the time. It was not until the nineties that interest in LCA really started growing, which is reflected in the growing number of publications, workshops and congresses.

In this surge of activity, the Society of Environmental Toxicology and Chemistry (SETAC) plays a leading role. Through its North American and European branches, it has organized an extremely large number of workshops, bringing together researchers from Europe and the USA, particularly, to discuss a wide variety of LCA issues. The International Standards Organization (ISO) is also active in the field of LCA.

The most important result of SETAC's activities to date is the elaboration of a preliminary

version of a "Code of Practice". One of the key elements of this "Code of Practice" is the methodological framework, which consists of four components: goal definition and scoping, inventory analysis, impact assessment (with further subdivision into classification, characterization and valuation) and improvement assessment.

Work on this doctoral thesis was started in 1989, just prior to the surge of international activities. The thesis was written in interaction with the activities described above, particularly those of SETAC. On the one hand, SETAC has provided fora for presenting and discussing the results reported here. On the other hand, the same results have made an important contribution to discussions within the SETAC framework.

One such contribution is described in Chapter 2: the definition of a general methodological framework for LCAs, and a detailed description of the first two components of this framework.

In 1989, when work on this chapter was started, but also partially today still, the term LCA encompassed a wide variety of methods, each with its own shortcomings. This frequently gave rise to a confusion of tongues. The aim of this methodological framework, then, was to structure discussions on the various LCA methodologies and their shortcomings. The framework comprises five components: goal definition, inventory, classification, valuation and improvement analysis.

The main difference between this framework and that elaborated by SETAC (in the "Code of Practice") is that in the SETAC framework classification and valuation do not constitute separate components but are steps within the impact assessment component, and that the classification component as described in this thesis is subdivided in the SETAC framework into classification and characterization. Further differences are merely terminological.

The reason for the differences between the two frameworks is historical in nature. The framework described in this thesis was elaborated before the SETAC framework. In fact, the framework in this thesis served as an input to the SETAC framework, undergoing several minor changes in the process.

In this thesis, on some points the subdivision of the framework into components has been taken a stage further. Most components are now divided into steps. Next, possible methods are discussed for each component and step, with proposals for improvement being presented in some cases. Some of these proposals have been elaborated in detail, while others have merely been roughly sketched. In Chapter 2 the goal definition and inventory are elaborated.

In the goal definition the goal of the study is defined. The goal is dependent on the application envisaged: is the aim to improve a product, to develop a new design, or to disseminate environmental data on one or more products, or is the study being performed in the context of an ecolabelling application, and so on. The kind of application is of influence on the entire procedure, determining how and how often the various methodological components are to be run through. All applications involve some form of comparison of products, and a basis for comparison must therefore be established. In this, the function fulfilled by the products being compared plays a key role, and for this reason the unit of comparison is frequently termed the "functional unit". As an illustration, it is incorrect to compare 1 pack of milk with 1 bottle of milk, since the former is used only 1×, while the latter can be used 25×. A better basis for comparison is the packaging of 1000 litres of milk, which requires 1000 packs or 40 bottles. In the goal definition,

finally, explicit mention should be made of the spatial scale (the Netherlands or Europe, for example) and the time period (are the packs and bottles today's, yesterday's or tomorrow's?).

In the inventory, the life-cycle constitutes the guiding principle. In order to obtain a quantitative picture of the environmental interventions of a product system, the boundaries between the product system and the (natural) environment and those between the product system and other product systems must be established. In establishing the boundaries between the product system and the environment the following kinds of question arise: is a production forest part of the environment or is it an economic process within the product system that requires fertilizers, pesticides and space?; does waste disposal belong to the environment or is it an economic process requiring space, causing emissions to the air, water and soil and perhaps producing methane that can be used a source of energy? In this thesis it is argued that forestry and waste disposal are economic processes with their own particular environmental interventions and that they should therefore be treated in an LCA.

In establishing the boundaries between the product system under study and other product systems, the problematical issues centre around three different situations: production of coproducts, combined waste processing, and open-loop recycling. In all three cases, the processes of the product system studied create not only an economic output to that system, but also at least one economic output to another product system. The environmental interventions of that specific process must therefore be allocated in some way to the various product systems. Allocation in proportion to a physical unit such as the mass of the economic outputs concerned is a simple approach, but the results of this type of allocation are highly dependent on the unit chosen (kg or mole, for example). Although allocation in proportion to the economic value reflects well the value assigned by society to the economic outputs in question, this approach requires additional data on the prices of the outputs.

Behind every economic process are various other economic processes. In the inventory it must be determined how to cut off this, in principle infinite, regression of processes (with one being required for the other) within a product system. It appears virtually impossible to find golden rules for such a decision.

The next issue also demands a well-argued choice. For each process a choice from among several alternatives is required, since there is generally more than one technology available for making the same product. In making a choice from among such processes, it is proposed to work with average, recently installed technologies. In addition, this choice depends on the scale of the comparison in question (a single facility, the Netherlands, Europe, etc.).

The result of the inventory is termed the inventory table: a list of environmental interventions such as resource extractions, and hazardous emissions to the air, water and soil.

A comparison of a number of completed studies of milk packaging showed that these differed in many respects. For example, the functional unit, and particularly its specification, was different in each study, different definitions were given of what were essentially the same product system, the choice of processes and data differed, and so on. The various studies consequently gave extremely divergent results.

In Chapter 3 the other three components of the aforementioned methodological framework are discussed in detail: classification, valuation and improvement analysis.

In classification the contribution of the environmental interventions calculated in the inventory to a number of generally recognized environmental problems (environmental themes) are quantified. The result is sometimes termed the environmental profile of a product, and consists of a number of so-called effect scores. In developing the classification component, four steps can be distinguished: 1. definition of generally recognized environmental problems that deserve consideration in an LCA; 2. definition of classification factors (also known as equivalency factors) representing the contribution of one quantified unit of an environmental intervention to a given environmental problem; 3. multiplication of the interventions by their equivalency factors and aggregation of the results of this multiplication into a number of effect scores; and 4. "normalization" of these effect scores.

Eighteen environmental problems are distinguished, and these can be grouped together into three categories: depletion, pollution and disturbance. Whether all these problem types will be included in a particular case study depends on the availability of inventory data and equivalency factors. In addition, an effect score may of course be zero if the product system simply does not contribute to the problem in question.

In this thesis proposals are presented for equivalency factors that until recently were entirely lacking. For pollution problems such as global warming, ozone layer depletion, photochemical smog formation, acidification, human toxicity, ecotoxicity and eutrophication, equivalency factors have been drawn up for as many different substances as possible. In achieving this aim, for a number of problems such as global warming, ozone layer depletion, photochemical smog formation and acidification, "thorough scientific shopping" proved adequate. This process yielded, respectively, the Global Warming Potential (GWP), the Ozone Depletion Potential (ODP), the Photochemical Oxidant Creation Potential (POCP) and the Acidification Potential (AP) as equivalency factors. For other types of problems ready-made methods were lacking, and new methods therefore had to be developed. For depletion and disturbance problems, too, ready-made equivalency factors proved scarce. In this thesis proposals are presented for some of these equivalency factors that are still lacking. For example, in Chapter 4 a proposal is made for quantifying the effects of toxic emissions, and in Chapter 5 for quantifying the effects of resource use. The main focus of this thesis is thus on the classification component.

Despite these proposals, there is a need for continued research on this point, as well as for improving a number of already existing equivalency factors, preferably in specialized scientific debating fora, whether existing (for global warming, for example, the Intergovernmental Panel for Climate Change, the IPCC) or yet to be created.

In order to gain a better impression of the significance of the various effect scores, these can be divided by the total magnitude of the problem in question, expressed in the same terms. This step is termed normalization and can be used to obtain an idea of the degree to which the product system under study contributes to the overall magnitude of the environmental problems. In this way, differences on this point between the various effect scores can be identified.

The valuation component consists of an evaluation of the effect scores of the environmental profile and an assessment of the reliability and validity of the results of the overall study. Societal values and preferences play a key role in valuating the effect scores. First, it is assessed whether one particular product alternative is superior or equal on all criteria compared with the other alternatives. If this is the case, further weighting is unnecessary. If an unweighted valuation

of this kind does not lead to a clear-cut result, as is often the case, it will be necessary to assign weighting factors to the effect scores - at least if a conclusive result is desired. Whether this is indeed the case depends on the goal and application of the study. This weighting can be carried out by means of qualitative or quantitative multi-criteria analysis, making use of expert opinion or a standard list of weighting factors. Quantitative multi-criteria analysis based on a standard list of weighting factors that is adjusted annually, for example, is quick, repeatable and easy to apply. These represent important advantages in assessing the environmental performance of products, since there are already millions of products on the market, with many new ones appearing daily. For the purposes of LCA, then, this kind of formalized weighting procedure would appear to be the best. The greatest problem here is to obtain a standard list of weighting factors that enjoys broad public support.

There is little point in valuating environmental profiles if the reliability and validity of the data have not been duly assessed. Methods for this are still largely lacking, which is due partly to the lack of confidence intervals in most datasets. For such cases, Heijungs has proposed applying a marginal analysis to track down those data which have the greatest influence on the end result and for which maximum accuracy should be aimed for. If the credibility of LCA studies is to be improved, greater efforts must certainly be devoted to this aspect.

In the improvement analysis options for improvement are identified. At the moment, two complementary techniques are available for this purpose: dominance analysis and marginal analysis. These two techniques permit identification of a number of points of departure for improving the product systems under study. Subsequently, technological, financial-economic and other knowledge outside the scope of an environmentally oriented LCA will have to be applied to assess whether or not individual improvements are feasible. With the aid of the aforementioned techniques and supplementary knowledge from outside the field of the environmentally oriented LCA, LCA can become an analytical tool for environmentally oriented design and thus encourage continual improvement of products.

As stated earlier, in Chapter 4 a new method is proposed for defining equivalency factors for human toxicity and ecotoxicity.

To define proper equivalency factors for human toxicity and ecotoxicity is one of the most difficult problems in the classification component of LCA. Until now, the aforementioned critical volume approach has been used in practice. As explained earlier, in this approach emissions of potentially toxic substances to the air and water are aggregated into so-called critical volumes of air and water, respectively. In doing so, exposure routes and environmental processes are left out of consideration; impact assessment is based on semi-political standards for the substances in question. This approach might be improved by developing equivalency factors for human toxicity and ecotoxicity in two stages: an exposure part, in which substance emissions are converted into concentrations to which a receptor may be exposed, and an effect part, in which concentrations are related to effects via the threshold value (threshold concentration) below which the receptor suffers no adverse effects. Both elements would have to be determined for each individual substance and be based as far as possible on scientific models and empirical data. This would also seem to be a realistic objective.

For both human toxicity and ecotoxicity the exposure part might, for example, be

calculated using the so-called multimedia models developed by Mackay. However, such multimedia models cannot be applied directly to the emissions as quantified in the inventory, because these emissions are not specified with regard to the dimension of time. The substance emissions computed in the inventory take place at an unknown rate. In an LCA, emissions are related to a function, and as a result the time dimension is divided out. LCA emissions are therefore pulses (in kg). Multimedia models incorporate time-dependent processes such as degradation and transport; these models work with emissions as fluxes (in kg/day, for example). One possible solution to this flux-pulse problem is to choose a reference substance and then compute a dimensionless equivalency factor for each substance, analogous to the concepts of ODP, GWP and POCP.

In elaborating the effect part for human toxicity, due allowance must be made for the many mechanisms and many types of effect, and also for carcinogenic and mutagenic substances for which no threshold value can be derived. In this thesis it is proposed to resolve these problems by taking the lowest threshold value - in other words the first concentration at which an effect is observed - and, for carcinogenic and mutagenic substances, by deriving virtual threshold values by defining "acceptable" elevated risks. In this way a so-called HTP (human toxicity potential) can be calculated for each substance. This HTP is an expression of the human toxicity of an emission of a given substance relative to the human toxicity of an equal emission of a reference substance.

Elaboration of the effect part for ecotoxicity means solving problems parallel to those for human toxicity. In assessing ecotoxicological effects, moreover, some way must be found to allow for the multitude of species in ecosystems. One solution to this problem can be formulated in the same way as for human toxicity, although it is now also necessary to derive threshold values for ecosystems from toxicity data for individual species. There are a number of methods available for this purpose, and others are under development. In this chapter it is proposed to distinguish between terrestrial and aquatic ecosystems, moreover, because of the differences in the species found in these two media and because of the different pathways along which these species are exposed to toxic substances. As specific toxicity data for ecosystems in aquatic sediment and for exposure via the atmosphere are lacking, these two compartments have provisionally been omitted from consideration. Using the method proposed here, a so-called TETP (terrestrial ecotoxicity potential) and an AETP (aquatic ecotoxicity potential) can be calculated for each substance. The TETP and the AETP express the terrestrial and aquatic toxicity, respectively, of an emission of a given substance relative to the terrestrial and aquatic toxicity, respectively, of an equal emission of a reference substance.

It should be stressed that the HTP, TETP and AETP approaches are still in an early stage of development and that concrete values have not yet been derived for a single substance. One of the obstacles that needs to be overcome is the limited applicability of multimedia models. As they currently stand, these models can only be applied to most organic compounds, but not to metals, metallic compounds, salts or polymers. Although attempts are being made to resolve these problems, considerable research is still required before the improved models can be applied in practice. Another major problem is the limited availability of the substance data required for multimedia models. Although this data problem can in principle be solved, progress in this area is extremely slow.

As mentioned above, in Chapter 5 of this thesis a proposal is elaborated for defining

equivalency factors for resource depletion. In this chapter it is proposed to make a distinction between depletion of biotic and of abiotic resources. Equivalency factors for resources contributing to one of these problems are calculated on the basis of physical data on reserves and resource deaccumulation. Deaccumulation is defined as the annual production (extraction) minus the annual regeneration of the resource in question. In this chapter a number of equations are elaborated for calculating the equivalency factors from these data. This leads to definition of so-called abiotic depletion potentials (ADP) and biotic depletion potentials (BDP). Subsequently, ADPs are calculated for virtually all the elements of the periodic table. ADPs for compounds can be derived from the ADPs of the constituent elements by determining the ratio of these elements in the compound and multiplying that ratio by the ADPs of the individual elements.

The resource classification method presented in this thesis is only one of many possible methods. In this context, it is interesting to mention that there is a remarkable convergence between the method elaborated here and the methods proposed at the LCA workshop held in Sandestin, USA from 1 to 7 February 1992. The nature of the depletion problem makes it virtually impossible to establish "the correct" method, because in the case of depletion correctness cannot be empirically validated. This holds both for the definition of the abiotic depletion problem and for the choice of the mathematical equation. For the abiotic depletion problem there are at least two possible definitions. The abiotic depletion problem can be defined as the decreasing availability of primary natural resources, or as the decreasing availability of resources in general. In the latter case, resources that can, for example, be recovered from waste - with however much effort - also count as reserves. In this thesis the former definition has been taken as the point of departure. If the latter definition were to be employed, it would be useful to add entropy to the list of data required. The choice of the "correct" mathematical equation for calculating equivalency factors is also problematical, if not impossible. The power to which the various parameters are raised in this equation is to a certain extent arbitrary.

Finally, it should be remarked that the most desirable dataset for resource reserves, *viz.* the quantity that can ultimately be technically (not economically) recovered from the earth's crust, the sea and the atmosphere, will, by definition, never be available. Consequently, a second best dataset will always have to be used.

In Chapter 6 of this thesis a methodological case study has been elaborated. This case study was carried out in order to obtain an impression of the degree to which results are affected by the choice of methodology and data in the various components and steps. The case study also serves to illustrate the various methodological proposals. Four different margarines that can be used for frying, roasting and spreading on bread have been evaluated and compared. The objective was not to arrive at a definitive final judgment; the study was carried out purely as a methodological exercise. To determine the influence of the choices of methodology and datasets on the final result, the following aspects have been subjected to a sensitivity analysis: estimates of pesticide emissions, impact assessment methods, valuation methods, coproduction allocation methods, resource classification methods, and datasets for Global Warming Potentials and Photochemical Ozone Creation Potentials.

First, results are calculated on the basis of a number of default options for the aforementioned choices. In doing so, a major error in the data on pesticide emissions came to

light. The precise magnitude of this error is unknown, but is estimated to be a factor 1000. In the results, this error is completely predominant; if a more realistic estimate of pesticide emissions is used, an opposite picture is obtained.

The sensitivity analyses showed that the results are largely dependent on the assumed distribution of the pesticide emissions over the various environmental compartments (soil, water, air). Both the estimate of pesticide emissions and the assumed distribution of these emissions will therefore have to be considered in greater depth before any useful final evaluation of the various margarines can be made.

The results were also found to be dependent on the choice of impact assessment method. Three methods were compared: the multi-step method described in this thesis (also referred to as the "environmental themes" approach); the single-step EPS method, developed in Sweden; and the single-step Ecopoints method, developed in Switzerland. The differences between the results of the three methods are significant, which is not surprising given the very wide range of the emissions considered (the EPS and Ecopoints methods do not allow for assessment of pesticide emissions, for example) and the similarly wide range of the weights assigned to these emissions (on the basis of environmental as well as other considerations).

With the impact assessment method described in this thesis (the "environmental themes" approach) there is an additional source of uncertainty, which is related to the choice of equation for the so-called sustainability factor in the valuation. The sustainability factor provides an indication of the degree to which the current level of environmental impacts differs from a sustainable level. In this methodological case study, one method has been elaborated to arrive at a standard list of such sustainability factors. Given a number of basic constraints, the equation with which current and sustainable levels are weighed against one another is an open choice. In this case study, the results obtained with two different equations were compared: the outcomes differed only marginally.

The choices that can be made with respect to coproduction allocation methods, resource classification methods and datasets for Global Warming Potentials (GWP) and Photochemical Ozone Creation Potentials (POCP) were found to have little influence on the final results. Regarding resource classification methods and GWP and POCP datasets, the final results differed only marginally in an absolute sense. For coproduction allocation methods, the situation is different. The sensitivity analysis carried out showed that, for all environmental problems, there may be substantial differences between the effect scores of the various margarines. For all four margarines, however, the changes are of about the same magnitude, which is due to the fact that the process trees of the four margarines are extremely similar. As a consequence, the choice of coproduction allocation method, too, has little influence on the final results.

The nature of these results might give one the impression that the results of an LCA are entirely dependent on the choices made. Although this impression can never be removed completely, because methods can also be abused, it is to be anticipated that the range of results will diminish with time. The largest margin in the results of this case study is due to the quality of the pesticide data. These errors can be avoided by improving assessment of data quality and by peer reviews. Generally speaking, greater attention should be paid to the quality and availability of the data required for LCA. This is especially true for the inventory, but certainly also for the classification and the valuation. There is at the present time a real danger of LCA losing (more of)

its credibility because the data used are incorrect, or incomplete on certain key points.

In Chapter 7, finally, the main findings of this thesis are discussed, component by component and step by step of the methodological framework. Since 1989 major steps forward have been made in developing LCA methodology. In this context, the main contributions of this doctoral thesis are the proposal for the methodological framework, the problem-oriented (theme-oriented) classification scheme and the proposals for a number of specific equivalency factors. The proposed methodological framework has been discussed in the SETAC setting and adopted in modified form (see above). The problem-oriented classification scheme and the proposals for equivalency factors are still the subject of international scientific debate.

Not all methodological problems have yet been solved, however. To the extent that these problems are related to the choices and assumptions that must be made in the course of an LCA study, these problems will never be able to be "solved" but, with each new study, a reasoned explanation will have to be given of the approach adopted. This is particularly valid for the goal of the study, for example, and in particular for the functional unit, but also for the choice of the system boundaries, the representative technology, the environmental problems being considered and the methods employed for quantifying and weighting the contributions of the product systems under study to these environmental problems. Other remaining methodological problems could be solved by conducting further research. At the end of this thesis, a number of suggestions for further study are made, concerning the speed of the method, improvement of methodological elements and improvement of the quality and availability of data.

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Curriculum vitae

Jeroen Bartholomeus Guinée werd op 27 januari 1962 geboren te Utrecht. In 1980 behaalde hij het diploma gymnasium ß aan het Bonifatius college te Utrecht. Hij studeerde milieuhygiëne aan de Landbouw Hogeschool te Wageningen van 1980 tot 1987. Het kandidaatsexamen legde hij af in mei 1985. Tijdens de doctoraal-fase waren de hoofdvakken Agrarisch Recht en Waterzuivering. Daarnaast is een stage-periode vervuld aan de Universiteit van Aveiro te Portugal en is nog 3 maanden gewerkt in een studiegroep "Marktconforme Beleidsinstrumenten" aan het Centrum voor Milieukunde van de Rijksuniversiteit Leiden. In deze studiegroep is de grondslag gelegd voor één van de huidige onderzoeksspeerpunten van de Sectie Stoffen & Produkten van het Centrum voor Milieukunde: de stofstroomanalyse. Na een kortstondige onderhouding door de G.S.D van de gemeente Wageningen, vervulde hij zijn vervangende dienstplicht aan hetzelfde Centrum voor Milieukunde van 1987 tot 1989. Daar heeft hij in de beginperiode meegewerkt aan een aantal projekten op het gebied van de stofstroomanalyse en op het gebied van de levenscyclusanalyse van produkten. Van 1 september 1989 tot 1 mei 1994 heeft hij aan dit proefschrift gewerkt, met een onderbreking van 7 maanden waarin hij enige projektleiderstaken heeft vervuld. Het werk aan het proefschift vond plaats aan het Centrum voor Milieukunde onder leiding van prof. dr. H.A. Udo de Haes en dr. G. Huppes. Jeroen B. Guinée werkt nu als coördinator van een vijf-jarig interuniversitair metalenprojekt op het gebied van de stofstroomanalyse gefinancierd door NWO en als projektleider van een aantal kortlopende projekten aan het Centrum voor Milieukunde.